APPENDIX A
RECEPTOR PROFILES

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SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
Site	San Jacinto River Waste Pits Superfund site
USEPA	U.S. Environmental Protection Agency

1 INTRODUCTION

Receptors were selected for the baseline ecological risk assessment (BERA) to represent species of potential concern and the range of feeding guilds expected to inhabit terrestrial and/or aquatic habitat types at the San Jacinto River Waste Pits Superfund site (Site). Patterns of behavior, life history, and habitat use that affect the frequency and magnitude of exposure of each receptor to hazardous substances at the Site are quantified to estimate exposures for the ecological risk assessment. Quantitative estimates of parameters describing rates of ingestion of soil and food; information on life history (e.g., timing of migration and breeding); and habitat areas for each receptor are used to build exposure models. Specific information about how receptors use habitats at the Site can also be used to interpret the ecological significance of estimated exposures relative to effects thresholds.

Receptor surrogates for evaluation in the BERA for the northern impoundment were selected in the screening level ecological risk assessment (Appendix B to the Remedial Investigation and Feasibility Study Work Plan) as follows:

- Benthic macroinvertebrate communities
- Fish
  - Gulf killifish (*Fundulus grandis*)
  - Black drum (*Pogonias cromis*)
  - Southern flounder (*Paralichthys lethostigma*)
- Reptiles
  - Alligator snapping turtle (*Macrochelys temminckii*)
- Piscivorous birds
  - Neotropic cormorant (*Phalacrocorax brasilianus*)
  - Great blue heron (*Ardea herodias*)
- Invertivorous birds
  - Spotted sandpiper (*Actitis macularius*)
  - Killdeer (*Charadrius vociferous*)

- Semiaquatic mammals
  - Raccoon (*Procyon lotor*)
  - Marsh rice rat (*Oryzomys palustris*).

Aquatic and terrestrial invertebrates are not discussed in this Appendix. Only fish, reptiles, birds, and mammals are discussed below because specific life history or biological factors affect exposures in a manner relevant to risk estimation for these receptors. Information on the proposed receptors for the south impoundment is also discussed herein (killdeer, pocket gopher [*Geomys breviceps*], common garter snake [*Thamnophis sirtalis*]; see also Appendix E) and will be used in the BERA for the south impoundment. Quantitative biological variables required for modeling the exposures of fish, alligator snapping turtle, birds, and mammals included:

- Body weight (kg)
- Average home range (km or km^2)
- Rates of ingestion of food, water (for reptiles, seabirds, and wading birds) and soil or sediment (kg diet/kg body weight [bw]-day)
- Composition of the diet.

Because surface water in the vicinity of the Site is brackish, wildlife other than reptiles, seabirds, and wading birds are not expected to ingest water at the Site, so water ingestion is not modeled as part of the exposure estimation for mammals or terrestrial birds. However, seabirds such as cormorants have nasal glands that allow them to concentrate and excrete salts from the blood following ingestion of saline waters, while wading birds like sandpipers have been shown to tolerate ingestion of water at low salinities (Purdue and Haines 1977) such as are present at the site.

Other variables that can be important to make risk models more realistic or that support qualitative interpretation of estimated exposures relative to effects levels include:

- Numbers of breeding cycles per year
- Numbers of young per brood and period of gestation
- Seasonal patterns of migration and months expected on site
- Seasonal changes in diet

- Preferences for certain habitat types or physical conditions
- Density and territoriality.

To compile the necessary information, relevant descriptors of each receptor were taken from primary scientific publications, from synthetic reviews *(e.g., Wildlife Exposure Factors Handbook*; USEPA 1993), and from ecological risk assessments conducted in U.S. Environmental Protection Agency (USEPA) Region 6 (e.g., Patrick Bayou Superfund Site). Available data include a range of studies from across North America and Europe conducted in a variety of climates and habitat types. In many cases, only one or two studies were available to describe a species of interest. When multiple values for a relevant exposure parameter were found in the literature, the following were considered in the final selection of a quantitative value:

- Geographic proximity to the Site
- Ecological similarity to the Site (e.g., southern temperate estuarine and mixed-deciduous riparian systems)
- Climatic similarity to the Site (e.g., southern temperate ecosystems).

The following sections provide the basis for specific exposure assumptions used in the exposure modeling. Detailed exposure assumptions are presented in Section 4 of the main text of the report. Receptor profiles also provide context for interpretation of risk models.

2 FISH

Three fish species found at the Site are used as receptor surrogates for the ecological risk evaluation: Gulf killifish, black drum, and southern flounder. These species were selected because they incorporate several life history characteristics important to measuring exposure to bioaccumulative contaminants, including long lifespans, limited home ranges (a focus on non-migratory species), proximity to sediments and feeding from the benthic environment, and mid- and upper-trophic level diets. They also represent important prey species for upper trophic levels. Relevant information on each species is summarized below.

2.1 Gulf Killifish

The Gulf killifish is a relatively small (up to 18 cm), omnivorous, euryhaline fish that is commonly found in estuaries, tidal marshes, bay shores and river reaches of the Gulf Coast. Native to the western Gulf of Mexico slope drainages, the Gulf killifish, often called the "bait minnow" in Texas, inhabits drainages along the coast from the northeastern coast of Florida to the Gulf Coast of Texas and Cuba (Schofield and Fuller 2011; Hassan-Williams et al. 2011). While these fish tolerate a wide range of salinities, they are reported to favor open, brackish waters to inland ponds (Hassan-Williams et al. 2011). The Gulf killifish is among the most abundant fish species in vegetated areas (inner *Spartina alterniflora* or *S. patens* marsh) of upper Galveston Bay in the spring and fall (Rozas and Zimmerman 2000, as cited in Hassan-Williams et al. 2011).

The Gulf killifish feeds throughout the water column, preying on grass shrimp (*Palaemonetes*), microcrustaceans (copepods), mosquito (*Dipteran*) larvae and pupae, and small fishes, terrestrial insects on the surface, and bivalve molluscs, benthic algae and aquatic plants (Schofield and Fuller 2011; Hassan-Williams et al. 2011). They have been observed to increase their feeding rate when gaining access to flooded marshes (Schofield and Fuller 2011).

In Texas, the spawning season for Gulf killifish is from March to October. The preferred habitat for spawning is shallow areas within dense beds of marsh grass, clumps of debris, or among oysters. Eggs are deposited on vegetation during periods of maximum high tides (spring tides), develop while they are exposed to humid air, and hatch when inundated on

the next extreme high tide, usually in about 11 to 15 days (Schofield and Fuller 2011; Hassan-Williams et al. 2011). Gulf killifish are non-migratory.

2.2 Black Drum

The black drum is a large-bodied, estuarine invertivore, found commonly in the bays and estuaries of the Gulf of Mexico. This fish is a member of the croaker family, so named because of its ability to produce croaking or drumming sounds with its air bladders. Younger, smaller fish, under a pound in weight, are referred to as "butterfly drum," while older drum of 30 pounds or more, of either gender, are called "bull drum." Black drum are widely distributed and abundant throughout Gulf waters, including bay, inshore, and offshore areas, and are among the most commonly caught sport fish and a mainstay in the commercial fishing industry (TPWD 2011a).

The black drum is found along the western Atlantic Coast from Nova Scotia south through the Gulf states to Mexico and along the southern Caribbean coast. It is a versatile species that can adapt to a wide range of habitats, from extremely warm clear waters found in shallow flats during the summer to Gulf waters at depths of more than 100 feet, and are known to survive better than many fish in freezing conditions (TPWD 2011a; Froese and Pauly 2009). They are attracted to freshwater creeks and rivers, can tolerate very salty waters, and have been found in turbid and muddy flooding sloughs.

The diet of the black drum consists primarily of molluscs, crab, and shrimp (TPWD 2011a). Black drum are very adaptable and thus are found year round in a very wide variety of estuarine and marine habitats along the Gulf Coast. Black drum are resident within localized embayments when forage is plentiful and water conditions are acceptable, but have been observed to migrate in search of food and more desirable habitats, as far as 245 miles in 1 year, though distances of less than 10 miles are more common. Spawning migrations are notable. Large drum, at least 4 to 5 years of age, school in deeper bays and channels before spawning, an event known as a "Bull Run" among anglers. Free spawning occurs in these bays and channels in Texas primarily in February and March, and in the Gulf through April, with some spawning occurring later, in June and July (TPWD 2011a; Hill 2005).

Larval black drum inhabit the surf and bay shorelines in March and April and the half-inch to inch-long juveniles are found in early summer in shallow muddy creeks, sloughs, and areas that provide structure and cover (Froese and Pauly 2009). Young drum reach about 6 inches in length in the first year, 12 inches by the second year, and about 16 inches by the third year (TPWD 2011a). Black drum continue to grow and gain about 2 inches per year until they reach about 20 years of age. Females, which are characterized as prodigious, multiple spawners, mature at age 4 to 6 years (FWRI 2010). This species has been known to live to almost 60 years (FWRI 2010). Most bull drum weigh between 30 and 45 pounds; the largest black drum taken in Texas by a sports angler weighed 78 pounds (TPWD 2011a). Juvenile black drum are prey to a wide range of estuarine piscivores, including spotted seatrout (*Cynoscion nebulosus*) and crevalle jack (*Caranx hippos*). Larger drum are eaten by sharks (FWRI 2010).

2.3 Southern Flounder

Southern flounder is a large piscivorous flatfish common in the Gulf of Mexico. It is euryhaline, demersal (living and feeding near the bottom), cryptic (camouflaged) and resides over mud bottoms in estuaries and in coastal waters up to 40 m in depth (Froese and Pauly 2009). The southern flounder is found in the western Atlantic, ranging from North Carolina to the Texas coast and southward into Mexico, though is absent from the southern Florida coast and usually found west of the Mississippi River (Froese and Pauly 2009; TPWD 2011b). Mature southern flounder migrate out to the open Gulf in the winter to spawn and return to the estuaries in early spring, where they reside within their chosen home range for the remainder of the year. This species, which is a highly prized commercial food fish, is also targeted by recreational anglers at inshore bridges, jetties and from small boats in marshes and tidal creeks (Froese and Pauly 2009). The southern flounder tolerates low salinities and is found frequently in brackish bays and estuaries, and even on occasion in fresh water (Froese and Pauly 2009), except in the winter when most adult flounder have moved out into the Gulf of Mexico to spawn.

As with other flounder species, the mature southern flounder is compressed laterally, with the dorsal side of the body pigmented, the underside pale, and both eyes present on the dorsal (left) side of the body. The flounder is well-adapted to its habitat in that both eyes in

adults are on the "up" side of the head and the fish can alter the pigmentation of the upper side of the body to match the surrounding environment. The absence of an air bladder, as well as its small body cavity, help the flounder stay close to, or lie on the bottom and wait to ambush prey (TPWD 2011b).

Juvenile flounder feed mostly on small benthic invertebrates such as crustaceans, and add fish increasingly to their diet as they grow (Hill 2005). Adult southern flounders are almost strictly piscivorous, but will opportunistically feed on large invertebrates (i.e., crabs and shrimp) (Hill 2005).

Female southern flounder reach maturity at about 2 years and, coincidental with a 4 to 5°C drop in water temperature in the fall, leave the bays for spawning in the Gulf of Mexico, trailing mature males who move a few weeks earlier (Hill 2005). In Texas, this migration occurs primarily between October and December. Spawning occurs at depths of 50 to 100 feet. The flounder eggs are buoyant and hatch within the Gulf. By late winter and early spring, the larval fish, whose eyes are located on opposite sides of the head and who swim in an upright position are passively transported by tides and currents through Gulf passes into tidal estuaries and flats, where they settle to silty or muddy substrates to undergo metamorphosis. During metamorphosis, the right eye "migrates" to the left side of the head and the fish ultimately assumes its left-side-up position for life (Hill 2005; TPWD 2011b). Juvenile flounder then enter lower salinity waters of coastal rivers, creeks, and bayous. Small flounder grow rapidly and may reach 12 inches in length by the end of their first year (TPWD 2011b). Females grow faster than do males, reaching an average of 12 to 14 inches by maturity. Males average from 8 to 12 inches in length (SCDNR 2011). Adult southern flounders can grow as long as three feet and weight up to 20 pounds.

3 REPTILES

The alligator snapping turtle and the common garter snake were selected as reptile receptors for the BERA. Although it is expected to be very rare in the brackish waters such as those in the San Jacinto River estuary, the alligator snapping turtle was selected as a receptor surrogate because it spends most of its life in water and is omnivorous, consuming plants, small fish, insects, snakes, and carrion. The garter snake was selected because it is a common, invertivorous reptile whose habitat requirements overlap with the conditions present in the upland portions of the site, and it is used for evaluating exposure to reptiles in the south impoundment (Appendix E). Additional information on these receptors' life histories and feeding behaviors is provided below.

3.1 Alligator Snapping Turtle

The alligator snapping turtle, which is the largest freshwater turtle species in the world, resides in freshwater areas of the southeastern portion of the United States, ranging from northern Florida and southern Georgia through the Mississippi River Valley and Gulf states into eastern Texas and Oklahoma (Nichols et al. 1999). The most established populations of this turtle are found around large water bodies, such as the Mississippi River, but they have been found in a variety of environs including lakes, oxbows, bayous, deep rivers, canals, creeks, ponds and brackish estuaries (Franklin 2011). Although primarily a freshwater species, alligator snapping turtles may occasionally use brackish habitats; for example, an adult male was collected in upper Mobile Bay, Alabama, with brackish-water barnacles on its carapace, indicating it had spent sufficient time in saline waters for these organisms to attach to and grow on the carapace (Jackson and Ross 1971). Adult alligator snapping turtles are characterized by three large, pronounced ridges that run from the front to the back of their carapace. Like all snapper turtles, the alligator snapper head is large, its jaws are very powerful, and the turtle is unable to retract the head into the shells. While some captive specimens have reached very large proportions, males in the wild average 26 inches in shell length and weigh about 175 pounds and females are much smaller, with a maximum recorded weight of around 50 pounds (National Geographic 2011). Alligator snappers can live in the wild for 50 to 100 years, though the average life spans for males and females of this species are estimated to be 26 and 23 years, respectively (Nichols et al. 1999).

Alligator snapping turtles are opportunistic omnivores, feeding primarily on fish and other turtles, and less frequently on frogs, snakes, snails, worms, clams, crayfish, carrion, insects and occasionally aquatic plants. These reptiles spend most of their time in the water, crawling on the bottoms of the deepest areas within large rivers, canals, lakes, and swamps during the day; they are known to stay submerged for 40 to 50 minutes, coming to the surface only for air (Nichols et al. 1999; Fuller and Somma 2011). Turtles thermoregulate, using different water depths, seasonally to retain optimal body temperature. Daytime feeding in the depths occurs mainly through ambush of prey (Fuller and Somma 2011). These turtles are more active at night, when they hunt and scavenge the shorelines. Only the nesting female ventures on land, and can feed on smaller terrestrial mammals such as squirrels, muskrats (*Ondatra zibethicus*), nutria (*Myocastor coypus*), and even opossums (family *Didelphimorphia*) and raccoons (Nichols et al. 1999). The only known predators of adult alligator snapping turtles are humans. The eggs and hatchlings of this species are an important source of food for large fish, raccoons, and birds (Nichols et al., 1999).

Alligator snapping turtles mate in early spring in Florida and late spring in the Mississippi Valley. Eggs are laid 2 months after mating, in nest holes dug in sand located about 50 m from a water body. Clutch size varies from about 9 to 50 eggs, and success is highly variable, dependent on predation and ambient temperature. Incubation takes about 14 to 20 weeks, with hatchlings emerging in the fall. Alligator snapping turtles invest no resources on parenting. Once the eggs are laid and nests are secured, the adults return to the water. Juveniles look similar to adults, but do not reach sexual maturity until 11 to 13 years of age.

Alligator snapping turtles prefer deep, large water bodies and submerged cover (e.g., submerged logs, beaver dams, overhanging shrubs) and substantial overhead canopy. Adults prefer to remain close to these core submerged features. The average home range of alligator snapping turtles in Oklahoma is reported to be approximately 880 linear meters for females and about 480 for males. Juvenile alligator snapping turtles range further than adults, with a home range averaging about 1,000 linear meters.

3.2 Common Garter Snake

The common garter snake is one of the most abundant snakes in North America. Of the four subspecies of the common garter snake found in Texas, the Texas garter snake (*Thamnophis*

sirtalis annectens) is the only subspecies known to inhabit eastern Texas locations; Harris County is one of several upper Gulf Coast counties in which these snakes have been observed in the last decade (Cannatella and LaDuc 2011). Regional populations of common garter snakes across the continent are distinguished mostly by variation in color patterns. The adult common garter snakes range in size between 46 and 137 cm (18 and 54 inches), and weigh an average of 150 g. The males are smaller than females and the young, which are similar in appearance to the adults, are born at 12.5 to 23 cm (5 to 9 inches) long (Zimmerman 2002).

The adaptability and resilience of the common garter snakes are evidenced by their residence in a wide variety of terrestrial and semiaquatic habitats, including meadows, marshes, woodlands, hillsides, and suburban and urban areas where debris, rock walls, foundations, gardens and other features provide good cover. These snakes prefer moist, grassy environments such as is found near the edges of ditches, ponds, lakes, and streams (Zimmerman 2002). In Texas, these snakes are found primarily in lowland habitats, particularly in areas with standing or running water, but can also be seen in open or edge habitats (Cannatella and LaDuc 2011). While these snakes tolerate a broader range of temperatures than do most, they bask in the sun during the day, and convene in coiled masses during sleep or hibernation to retain body heat. Hibernation occurs in natural cavities, rodent or crayfish burrows, under rock piles, or in stumps.

The common garter snake eats a variety of prey, dependent primarily on whether it is appropriately sized for swallowing whole. The adult diet includes amphibians, fish, and insects. Juvenile garter snakes eat a greater proportion of earthworms and insects than do adults. Baby birds, mammals, molluscs, and other snakes are also taken as prey (Cannatella and LaDuc 2011).

Garter snakes mate in the spring, as soon as they emerge from hibernation, and are ovoviviparous, meaning they carry their young until birth. In the summer and fall, the females birth an average of 26 young. The mother snakes allow the young to be around them for several days after birth, but do not provide any care, protection, or nourishment. These snakes reach sexual maturity, and maximum size, at three to four years of age, though Zimmerman (2002) indicates that the average lifespan of common garter snakes is

approximately two years and that most common garter snakes probably die in their first year of life.

Common garter snakes are eaten by a wide variety of predators, including large fish, bullfrogs (*Lithobates catesbeiana*), snapping turtles, milk snakes (*Lampropeltis triangulum*), American crows (*Corvus brachyrhynchos*), hawks, great blue herons, raccoons, foxes, squirrels, and shrews. These snakes can harbor a parasitic nematode, which resides in their tails, causing a shortened tail (Zimmerman 2002).

4 BIRDS

Birds selected as receptors for the BERA include the neotropic cormorant, great blue heron, spotted sandpiper, and killdeer. These species were selected because they are expected to be present in the vicinity of the Site and represent a variety of life histories and feeding behaviors. Relevant information on each of these species is provided in the following sections. In addition, brown pelican (*Pelecanus occidentalis*), bald eagle (*Haliaeetus leucocephalus*), and white-faced ibis (*Plegadis chihi*) are discussed as they are state-listed species whose ranges include the Site vicinity.

4.1 Neotropic Cormorant

The neotropic cormorant is one of the smaller cormorants; adults weigh between 1.2 and 1.4 kg (Telfair and Morrison 2005). Neotropic cormorants are year-round residents in coastal Texas. These birds are tolerant of a range of climatic and environmental conditions and inhabit wetlands in fresh, brackish, or salt water. In coastal areas, this cormorant is associated with sheltered bays, inlets, estuaries, lagoons, and rocky outcrops. Key habitat requirements include water deep enough for diving and elevated perches in trees, shrubs, or other structures for nesting, roosting, and drying plumage after feeding. While the neotropic cormorant is capable of perching in trees or on posts, pilings, and even cables and wires, the posterior location of its short legs and its thick and laterally flattened ankles make this bird a clumsy walker on land (Telfair and Morrison 2005).

Although neotropic cormorants have been reported to occasionally consume shrimp and amphibians, their primary prey is fish smaller than 8 cm (King 1989). They appear to forage opportunistically (i.e., feeding on species that are most abundant rather than selecting specific species). Neotropic cormorants forage mainly by pursuit-diving and are the only cormorant known to plunge-dive in shallow waters (less than 2 m depth) (Telfair and Morrison 2005). Foraging area for this species is present at the Site, and the species is likely to roost nearby.

The neotropic cormorant breeds annually, producing clutches of between 1 and 4 eggs, and can reach sexual maturity by 1 year of age. There is little information available regarding home ranges for this species; post-breeding dispersal by juveniles ranges between fidelity to

natal area, to dispersion several tens to hundreds of kilometers from the breeding site (Telfair and Morrison 2005). This species is tolerant of all but close and disruptive human activities.

4.2 Great Blue Heron

The great blue heron is the largest member of the heron family in North America, with body weight of males averaging slightly greater than body weight of females. A mean value of 2.2 kg for both sexes was assumed for this BERA (USEPA 1993). The great blue heron is found in freshwater and nearshore marine habitats throughout North and Central America. Habitats for great blue herons include streams, creeks, lake margins, and estuaries, with shallow water (<0.5 m) and a firm substrate on which to wade. Nearby wooded cover (within a few kilometers) is important for nesting. The great blue heron is a year-round resident of coastal Texas.

The preferred prey of great blue herons is fish; great blue herons will also eat amphibians, reptiles, crustaceans, insects, birds, and mammals (Alexander 1997; USEPA 1993). When fishing, great blue herons require shallow waters (to 0.5 m) with a firm substrate. Great blue herons consume relatively small fish that can be swallowed whole; 95 percent of fish consumed by a Wisconsin population of great blue herons were less than 25 cm in length (USEPA 1993).

In some areas, herons defend feeding territories, but in other areas, they are opportunistic and lack fidelity to a particular feeding site (USEPA 1993). Adult herons tend to feed the same type and size of food to their nestlings as they consume themselves. Predation on herons is mainly on eggs and young. Predators of young great blue heron eggs include crows and ravens. Eagles, raccoons, and hawks are among the animals which prey on the young birds and occasionally even adults (UMMZ 2011).

Great blue heron nests generally consist of a stick platform over 1 m in diameter; the nests may be reused and expanded for multiple years. Only one brood, with an average clutch size of 3 to 5 eggs, is raised per year, although if the clutch is destroyed, the parents may produce a replacement clutch. Both parents incubate and feed the young. Chicks fledge at approximately 2 months (UMMZ 2011). During the breeding season, great blue herons are monogamous and colonial. Breeding colonies are generally close to foraging grounds; a study

of great blue herons in Minnesota lakes found the distance between nesting colonies and feeding sites to range from 0 to 4.2 km, averaging 1.8 km (USEPA 1993). The median flight distance to feeding areas reported in a separate study in Minnesota was 2.7 km (Custer and Galli 2002).

4.3 Spotted Sandpiper

The spotted sandpiper is the most widespread shorebird in North America (Oring et al. 1997). Adult males weigh approximately 38 g (range = 34 to 41 g), while the larger females average about 47 g (43 to 50 g) (Maxson and Oring 1980; USEPA 1993). Spotted sandpipers are relatively common winter residents in some of the local habitats around the Houston Ship Channel (Litteer 2009) and their foraging habitats are present at the Site.

The spotted sandpiper obtains much of its diet by probing or "mining" soft sediments along shorelines (USEPA 1993). This species is a generalist feeder and will occupy almost all habitats near water, including the shorelines of ponds, streams, and rivers, as well as meadows, agricultural areas, and forested areas. This species typically forages within 200 m of the shoreline (Oring et al. 1997). Spotted sandpipers are visual foragers and prey on all manner of aquatic and terrestrial invertebrates and occasionally small fish.

Spotted sandpipers are winter visitors to coastal Texas. No relevant home range or foraging range information was found for this species. An estimated foraging range of linear shoreline of 1,500 m for sanderlings, a similarly sized, invertivorous shorebird that is known to winter in coastal Texas, was used for exposure modeling, based on data for a wintering population in central California (Macwhirter et al. 2002).

4.4 Killdeer

The killdeer is a relatively large upland plover (average adult weight of 88 g, UMMZ 2011; average adult female weight of 101 g, Jackson and Jackson 2000) that feeds predominantly on terrestrial invertebrates (e.g., earthworms, beetles, grasshoppers, and other small invertebrates). The species is widespread in open areas (e.g., agricultural fields, lawns, golf courses) throughout North America and is nonmigratory across the southern United States,

including Texas (Jackson and Jackson 2000). It is known to be common year-round in the vicinity of the Site (Litteer 2009).

Killdeer are primarily terrestrial invertivores. Stomach contents from killdeer in Texas were reported to contain 98 percent animal matter, mostly worms and insects (McAtee and Beal 1924).

This species is remarkably tolerant of constructed disturbances, and nesting has been documented from construction sites, road shoulders, and graveled rooftops (Jackson and Jackson 2000). Average nesting territories of killdeer in Minnesota were relatively small (0.57 acres). Larger, year-round home ranges of approximately 15 acres were reported for a northeastern California population; nesting period home ranges were smaller (Jackson and Jackson 2000). Nesting in Mississippi occurs from mid-March through late July and involves multiple broods (Jackson and Jackson 2000). The use of this surrogate species would be considered protective of smaller home range bird species at the Site (e.g., sparrows, wrens) that likely eat a larger percentage of plant matter, as well as larger omnivores (e.g., crows), and would also be protective of terrestrial ecosystem-based carnivores (e.g., hawks) that likely have larger home and forage ranges.

4.5 Brown Pelican

The brown pelican inhabits coastal areas from central North America to the Northern coasts of South America. These large seabirds, measuring from 100 to 137 cm in length and weighing approximately 2 to 5 kg, are recognized by their long bills, large gular pouch, darkly plumed body, and large (2 m) wingspans. Male pelicans are 15 to 20 percent heavier than are females and their bills are about 10 percent longer than the female's. Brown pelicans are distinct from other pelicans in that they are the only truly marine species of the pelican family.

Brown pelicans are highly social throughout the year and, in the nonbreeding season, congregate on sandbars, pilings, jetties, breakwaters, mangrove islets, and offshore rocks and islands to roost at night and rest during the day, after foraging. Breeding colonies are made up of thousands of birds, and are usually located on small, isolated estuarine, barrier or

offshore islands where predation by terrestrial mammals and disturbance by humans is limited and where 30 to 50 km of a consistent foraging habitat is available (Shields 2012). In Texas, major breeding colonies are found on Pelican Island in Corpus Christi Bay and on Sundown Island in Matagorda Bay. Bird Island in Matagorda Bay, older spoil islands in West Matagorda Bay, Dressing Point Island in East Matagorda Bay, and islands in Arkansas Bay occasionally support smaller groups or colonies of breeding pelicans (TPWD 2012a). The Texas Colonial Waterbird Census (undated, as cited in Shields 2012) indicates that a brown pelican colony exists on Little Pelican Island in Galveston Bay. This species is commonly sighted, in all seasons of the year, in Upper San Jacinto Bay (Baytown Nature Center 2006). The home range of the brown pelican is limited to its foraging ground, which is generally no greater than 20 km from nesting islands. Outside of the breeding season brown pelicans in California have been observed up to 75 km from the nearest island (Shields 2012). For the purposes of the risk assessment, a home range was estimated by taking the foraging ground estimate of 20 km from the nesting island (Shields 2012) and considering that value as a radius of a circle around a nesting island that could be used for foraging, to calculate a home range area of 1,257 km^2.

Brown pelicans usually forage in shallow (<150 m) estuarine and continental shelf waters within 20 km of nesting colonies during the breeding season, and up to 75 km from nearest land during the nonbreeding season (Shields 2012). They often feed by plunging, from mid-flight, head-first into the water to retrieve prey, and primarily capture fish within the first few meters below the surface (Shields 2012). This pelican species feeds on small schooling fishes throughout its range; along the Gulf Coast menhaden and mullet are predominant in its diet (TPWD 2012a). One study along the Gulf Coast showed menhaden constituted 96 percent of the diet, with silversides, dolphinfish, and prawn contributing approximately 3, 0.8, and 0.3 percent of the brown pelican diet, respectively (Shields 2012).

The breeding season on the Texas coast lasts from March through June, with the peak breeding activities in April and May. On the Gulf Coast, nests are built on the ground, on mud banks and ledges and, in vegetation on islands covered with mangrove or other woody vegetation (TWPD 2012a). Incubation of eggs is shared between the parents; eggs are incubated under the bird's large webbed feet. Adult pelicans regurgitate predigested fish to feed hatchlings. By 3 to 4 weeks of age, the young learn to prompt adults to disgorge whole

fish, which the young can swallow whole. Fledging occurs by 11 to 12 weeks of age and sexual maturity is reached by 3 to 5 years of age. Pelicans are long-lived, to approximately 30 years, and have reached 43 years of age (Shields 2012).

After decades of population declines stemming from exposures to organochlorine pesticides, the brown pelican was placed on the Federal Endangered Species list in 1970. Brown pelican reproduction subsequently improved and this species was removed from the Endangered Species List in the southeastern United States in 1985 and its population was thought to be restored along the Gulf coast by the late 1990s (Shields 2012). However, human disturbance and loss of nesting habitat continue to threaten the recovery of the brown pelican in Texas and this species still listed as "endangered" by the State of Texas (TPWD 2012a).

4.6 Bald Eagle

The bald eagle, easily recognized as an adult by its white head, dark brown plumage, and white tail, is the second largest bird of prey in North America, ranging in mass from 3.0 to 6.3 kg, with a wing span of 168 to 244 cm. Female bald eagles are 25 percent larger than are males, and both genders are smaller in the southeastern and southwestern regions of the United States than they are in northern climates (Buehler 2012). Bald eagles are opportunistic foragers, preferring to scavenge prey (carrion) or steal food from other species, but they can and will capture their own prey if these other sources of food are not available. The bald eagle prefers fish, but feeds on a variety of aquatic and terrestrial mammals, reptiles, amphibians, crustaceans, and a variety of birds, including waterfowl, gulls, and even great blue herons. A review averaging 20 studies across the bald eagle's range characterized the diet as approximately 56 percent fish, 28 percent birds, 14 percent mammals, and 2 percent other (Buehler 2012). The Texas Parks and Wildlife Department lists American coots, catfish, rough fish, and soft-shell turtles as the most common components of the bald eagle diet in Texas (TPWD 2012b).

In Texas, there are two populations of bald eagles: the breeding birds occur in the eastern half of the state and in coastal counties from Rockport to Houston; and, the wintering populations are found in the Panhandle, Central and East Texas, and in other areas where suitable habitat exists. While the majority of bald eagles observed in Texas are wintering

birds that breed in northern states (TPWD 2012b), the breeding populations of bald eagles are said to be building along the Gulf Coast, in Louisiana and Texas (Buehler 2012). San Jacinto county is listed among the 47 counties in which bald eagle nests had been known to occur as of 2003 (TPWD 2012b). However, the Baytown Nature Center (2006) census data reports that this bird has been observed only in the winter months, and only rarely, in Upper San Jacinto Bay.

Information on home ranges of the bald eagle varies widely, depending on breeding status of individual, season, and most importantly, food availability. Breeding adults have been observed to occupy from 7 to 22 km^2. Non-breeding bald eagles are nomadic, relative to breeding birds, and have been observed to occupy areas ranging from 10,000 to 55,000 km^2. (Buehler 2012). Wintering eagles' ranges vary based on whether or not individuals are associated with mates. Mated pairs range within hundreds of square km, while non-mated wintering individuals might range to 4,000 km^2. An average winter range of 310 km^2 was reported for bald eagles in Colorado, with mated pairs having a smaller home range (128 km^2) than unmated eagles (average home range of 547 km^2), while in Missouri, winter home ranges of 48 and 18 km^2 were reported over two consecutive years of study (Buehler 2012).

Preferred habitats of wintering bald eagles are characterized by proximity to abundant forage associated with open water and waterfowl habitats and by availability of desirable night roost sites, such as those afforded by the oldest, tallest trees that provide unobstructed views near water, on windbreaks, and in secluded canyons (TPWD 2012b). Wintering bald eagles often congregate in large numbers in habitats that provide adequate forage and roost sites that are buffered from inclement weather and human activity (Buehler 2012).

In general, the habitat of the breeding bald eagle is the same as that of the wintering eagle. Bald eagle nests are found in forested areas, adjacent or close (<2 km) to large bodies of water and other suitable foraging habitats, and removed from human activity. Bald eagle nests are rarely found at distances <500 m from human development (Buehler 2012). While eagle nests are constructed in a variety of tree species, preferentially in the tallest tree in an area, nests in East Texas are built primarily in loblolly pine (TPWD 2012b).

The nesting season in southern latitudes (e.g., Florida and other Gulf Coast states) is somewhat prolonged over those of northern climes, ranging from late fall through early spring (Buehler 2012). In Texas, eagles nest from October to July, with peak egg-laying occurring in December and hatching occurring in January (TPWD 2012b).

4.7 White-Faced Ibis

The white-faced ibis is a medium-sized wading bird, weighing from 450 to 525 g, standing about 2 feet tall, and identified at a distance by its long neck, legs and curved bill, and uniformly dark, maroon-brown plumage. During the breeding season, the white-faced ibis is distinguished by metallic bronze, purple and green sheens to the chestnut-maroon plumage, red legs, and a reddish purple face bordered by a thin line of white feathers separating the forehead from the face and extending around the back of the eye (Ryder and Manry 1994). The white-faced ibis is listed as a threatened species by the state of Texas (TPWD 2012c).

White-faced ibis occur mainly in the western United States, breeding in marshes and irrigation areas throughout the Great Basin, most commonly in Utah, Nevada, and California. The winter range of this bird is primarily coastal Louisiana and Texas south to several Mexican states, Guatemala, and Costa Rica (Ryder and Manry 1994). It is described as a year-round resident of coastal Texas and western Louisiana (Ryder and Manry 1994), though records from the vicinity of the Site indicate that it is more of an occasional spring/summer visitor in the Site vicinity. The Baytown Nature Center, located a few miles downstream of the Site on the San Jacinto River, lists this species as rare in winter and spring, and occasional in the summer and fall seasons (Baytown Nature Center 2006).

This species primarily inhabits freshwater wetlands and marshes, as well as swamps ponds and rivers, and is commonly found feeding in flooded agricultural fields, estuarine wetlands, and temporary, shallow wetlands created by rainfall or flooding (Ryder and Manry 1994). In Texas and Louisiana, the white-faced ibis nests mostly in coastal marshes and wetlands of the outer coastal plains (Audubon 2012; Ryder and Manry 1994). The preferred roosting habitats in Texas are low platforms of dead reed stems or on mud banks (Ryder and Manry 1994; TPWD 2012c). This species has also been observed nesting on bare ground in coastal areas dominated by the shrubby coastal plant sea oxeye (*Borrichia frutescens*) (TPWD 2012c;

Ryder and Manry 1994). The white-faced ibis breeds in large colonies at established roosting sites that are used repeatedly for several years, though changing water levels will prompt colonies to switch nesting locations. In Louisiana and possibly eastern Texas, the white-faced and glossy ibis species will breed in the same colony, though they do not interbreed (Ryder and Manry 1994).

In Texas, egg-laying and incubation extend from mid-April through early July, with three or four eggs hatching after an incubation period of approximately 21 days (TPWD 2012). The parents share in incubation and brooding activities. Fledglings leave the colony after about 6 or 7 weeks, usually accompanying adults to foraging grounds (Ryder and Manry 1994).

White-faced Ibis commonly feed in large flocks of as many as 1,000 birds or more. These birds wade in areas of shallow standing water, or traverse emergent ground, foraging for aquatic and moist-soil invertebrates. Prey on the water or ground surfaces are located visually, while tactile probing with their bills is used to find prey in sediment and soils. White-faced ibis prefer shallow (5 to 15 cm) wading depths and foraging areas with emergent vegetation (Safran et al. 2000). Prey items are usually rinsed in pools of water before being eaten (Ryder and Manry 1994), although the esophagi of birds collected in Nevada contained substantial amounts of soil (Bray 1986; Bray and Klebenow 1988). A variety of prey is taken, including insects, newts, leeches, small crustaceans, worms, fish, frogs, and snails (TPWD 2012c). Stomach contents of white-faced ibis collected in Louisiana most frequently contained crayfish and insect larvae, in addition to small fish, frogs, snails, small bivalves, and earthworms (Belknap 1957).There is relatively little information available regarding the home or foraging ranges of white-faced ibis. In Nevada, during the nesting period, most birds foraged 3–6 km (but up to 18 km) from the breeding colony; while breeding adults and recently fledged young ranged 40–48 km from colonies observed in Idaho (Ryder and Manry 1994). The territory size outside of the breeding period is unknown for this species. Based on the foraging ranges described by Ryder and Manry (1994) and expert opinion, the Great Basin bird observatory suggests a recommended habitat patch size of >1,200 ha, or > 12 km^2 (GBBO 2012).

5 MAMMALS

Two semiaquatic mammals are selected as receptors for the BERA: raccoon and marsh rice rat. These species were selected because they occupy different habitats and have different feeding behaviors and life histories. In addition, Baird's pocket gopher is a proposed receptor surrogate for terrestrial mammals to be evaluated in the ecological risk assessment for the south impoundment (Appendix E). Relevant information on each of these species is provided in the following sections.

5.1 Raccoon

The raccoon is the most abundant and widespread medium-sized, omnivorous mammal in North America (USEPA 1993). Raccoons exploit a wide variety of habitats. Habitats include floodplain forests, swamps, and marshes. Raccoons are extremely adaptable to human environments and can be found in abundance in suburban residential areas and farmlands. High-quality habitat for raccoons includes sites that have access to fresh water, trees or other structures for nesting, and high food availability including fruits, grains, invertebrates, and other animals.

Adult male raccoons in an Alabama study averaged 4.3 kg and adult females averaged 3.7 kg. In a Missouri study, male raccoons averaged 6.8 kg, and females 5.7 kg. Mortality is high in young-of the year raccoons; average lifespan in the wild is 5 years, with a maximum recorded age of 16 years (UMMZ 2011).

Raccoons are highly opportunistic feeders and omnivorous, with a diet that may include carrion, garbage, birds, mammals, fish, amphibians, reptiles, grains, fruits, most food prepared for human or domestic animal consumption, agricultural crops, and invertebrates, including insects, crayfish, and mussels. Proportions of different foods in the diet depend on location and season. Plant foods dominate raccoon diets for most of the year except during spring and early summer, concurrent with the breeding season, when animal matter may be consumed more frequently (USEPA 1993). Fish ranging in size from 2 to 9 inches (5 to 23 cm) were found in the stomachs of raccoons collected in Michigan by Alexander (1977). Food ingestion rates for raccoons were not found in the literature; an allometric equation for

placental mammals was used to estimate a daily ingestion rate for raccoon in the exposure model (Nagy 2001).

Raccoons escape many predators by remaining in a den during the day; they are alert and can be aggressive when active at night. Large predators may prey on raccoons, including coyotes, wolves, and owls, and their young may be taken by snakes (UMMZ 2011). Throughout most of North America, raccoons mate during February and March. Most females will produce one litter per year, and many raccoons produce litters within their first year of life. Gestation averages 63 days (Sanderson 1987), and most litters consist of three or four young. Nesting sites are primarily in hollow trees, but raccoons will also use ground dens, brush piles, and abandoned human structures for nesting, usually within a few to a few hundred meters of surface water.

Population densities are strongly dependent on habitat quality, including food availability and abundance of potential nest sites, with suburban areas generally having higher densities than rural/wild areas. Home range areas range from less than 0.05 km^2 in suburban neighborhoods to more than 5 km^2 in the wild, though values of one to a few km^2 are most commonly reported (USEPA 1993). Juvenile and adult males tend to have larger home ranges than do females (Sanderson 1987).

5.2 Marsh Rice Rat

The marsh rice rat is a semiaquatic, nocturnal, omnivorous rodent native to the southeastern U.S. This species typically inhabits marshy areas, but may be found anywhere that there is sufficient grass and groundcover to offer protection and foraging (Davis and Schmidly 1994). The marsh rice rat will readily use water to move among foraging areas and escape predation.

The marsh rice rat is considered an omnivore, with about equal parts animal and plant matter constituting its diet, though it may be more carnivorous in summer when animal prey is available (Sharp 1967). Leaves, seeds of marsh grasses and sedges, and fungus are part of the marsh rice rat's diet, and this species preys on a variety of animals including crabs, fish, insects, and bird eggs; they may occasionally scavenge carcasses of rodents and birds (Davis and Schmidly 1994). Sharp (1967) found a mixture of seeds and animal prey in stomachs of

rice rats captured on a coastal island in Georgia; animal prey consisted mainly of crabs and insects including dipterans and beetle larvae.

Marsh rice rats are sexually mature at 40 to 45 days old, and weight from about 40 to 68 g as adults (average weight of 51 g; Davis and Schmidly 1994; average adult female weight of 67.70 g ± 0.85 g [±standard error], Fernandes 2011). They can reproduce year round, and a female may produce five or six broods per year, consisting of two to seven offspring. Home ranges of 0.37 hectares for males and 0.23 hectares for females have been reported; average range lengths include 75 m for a Maryland population and 68 and 82 m for marsh rice rats in Florida (Wolfe 1982).

5.3 Baird's Pocket Gopher

The Baird's pocket gopher, also known as the Louisiana pocket gopher, is virtually indistinguishable, morphologically, from the plains (*G. busarius*) and Attwater's (*G. attwateri)* pocket gophers, each of which inhabit different regions of Texas (Sulentich et al. 1991; TPWD 2011c). These pocket gophers are small, dark brown, burrowing herbivores. With long, curved, and specially adapted front claws, a broad, flat head, tiny, bead-like eyes and rudimentary ears, and a compact body with skin and hair arranged to allow movement through borrows both backward and forward, these gophers are more highly specialized for digging than any other North American rodent (TPWD 2011c; KSR 2011; Sulentich et al. 1991). *G. breviceps* is the smallest of its congenerics, averaging 208 mm in length and weighing between 78 and 150 g, with an average reported weight of 100 g (MNH 2012). The Baird's pocket gopher is found in the eastern portion of Texas and has been found on both sides of the San Jacinto River in Harris County (Sulentich et al. 1991; TWPD 2011c).

Geomys live underground most of their lives and maintain labyrinths of burrows in sandy and loamy soils, digging to an average depth of approximately 6 inches and up to 2 feet, generally on treeless land (TPWD 2011c). As much of the burrowing is done in search of food, tunnels meander through feeding areas, and can extend well over 100 m. These rodents are solitary; each tunnel system is occupied by only one gopher. They rarely leave their burrows, except at night for mating or for limited foraging beyond the entrance (KSR 2011). In wet months, pocket gophers are known to live and nest in above-ground mounds

of dirt, in order to avoid being flooded out of their burrows and tunnels (Sulentich et al. 1991).

The Baird's pocket gopher is herbivorous, obtaining most of its food while digging tunnels and feeding primarily on underground roots and the stems of weeds and grasses. While most plant food is encountered and ingested while the gopher digs its lateral tunnels, green plants and grasses are obtained at night from around the entrance of the tunnels and beyond. Fur-lined cheek pouches are used to carry food and nesting material. Cellulose-digesting bacteria in the digestive system help the Baird's pocket gopher digest grasses and stored underground rhizomes during the winter and these gophers, as do many rodents, increase their utilization of food by re-ingesting their fecal pellets (Sulentich et al. 1991; TPWD 2011c).

The Baird's pocket gopher begins breeding in eastern Texas in early February and continues through August, with peak productivity occurring in June and July. One to four young are born to each litter (Sulentich et al. 1991). As with most rodents, the newborns are nearly naked, with eyes and ears closed, and are helpless at birth. The young remain with their mother until nearly full-grown, at about 6 to 7 weeks of age, when they disperse to lead an independent life (TPWD 2011c). Sexual maturity is reached within 90 days of birth (Sulentich et al. 1991).

In east Texas, Baird's pocket gophers are preyed on by long-tailed weasels, and, when caught out of their burrows, are vulnerable to king snakes (*Lampropeltis getula*), great-horned owls (*Bubo virginianus*), red-tailed hawks (*Buteo jamaicensis*), and striped skunks (*Mephitis mephitis*), among other common rodent predators (Sulentich et al. 1991; TPWD 2011). Because they remain protected in their burrows most of the time, pocket gophers are long-lived relative to many other rodents, living an average of 1 to 2 years in the wild (TPWD 2011c). The estimated population density in prairie habitat near College Station, Texas, was approximately 0.55 gophers per hectare (Sulentich et al. 1991).

5.4 Virginia Opossum

The Virginia opossum (*Didelphis virginiana)* is a widespread and adaptable nocturnal scavenger similar in size to a large house cat. (UMMZ 2003). It is the only marsupial found

north of Mexico. Opossums range from Central America through much of the continental United States, including the eastern two-thirds of the country and the coastal Pacific. Opossums range in size from 350 to 940 mm, averaging 740 mm. Adult males weigh an average of 5.5 pounds, and adult females average 4.0 pounds (Georgia DNR 2012); size may vary with location and climate (MNH 2012). The lifespan of a Virginia opossum averages 2 years, though many die in the first year of life (TPWD 2012d). Both northern and southern populations have white fur with black tips. They have a pointed snout, opposable thumb-like appendages and a scaly prehensile tail that can be used to climb, hang, or grasp objects (TPWD 2012c). The opossum is known for its tendency to "play dead": when exposed to a threatening situation, the opossum can enter a catatonic state in which its breathing nearly stops. The behavior is considered to be an involuntary defense mechanism (MNH 2012).

Opossums are well adapted to living near humans and occur in a variety of habitat types. They are primarily found in woodland areas especially near creeks, rivers, or lakes, but can also occupy marshes, farmland, prairies, and urban and rural environments. They prefer to live in hollow trees and logs, but can also nest under rocks, buildings, bridges, attics, woodpiles, or in other animals' abandoned burrows (UMMZ 2003; Georgia DNR 2012). In east Texas, Virginia opossums typically frequent overlapping home ranges approximately 0.05 km^2 in size, although the minimum size of home ranges may vary from 0.001 to 0.23 km^2. In East Texas woodland habitat, the density of opossums is about one opossum every 0.02 km^2 while in sandy, coastal parts of the state the density is about one opossum every 0.06 km^2 (Davis and Schmidly 1994).

The Virginia opossum has a brief gestation period of 2 weeks, after which the relatively undeveloped young crawl from the birth canal and attach themselves to the mother's nipple inside of her fur-lined pouch, where they stay attached for 7 weeks of nursing (UMMZ 2003). Litters usually consist of seven young, and Virginia opossums typically have two litters per year (Georgia DNR 2012).

Virginia opossums are omnivorous. Consuming mostly insects and carrion, the opossum also forages for acorns, berries, and other fruit and is also known to eat crustaceans, frogs, bird eggs and nestlings, small rodents, and the young of its own kind. In human-populated areas,

the opossum is known to scavenge for garbage and can be considered a nuisance for this reason (Georgia DNR 2012).

Common predators of Virginia opossums include canids, raccoons, and raptors. Humans are also a main cause of mortality through hunting and trapping, and opossums are frequently killed on roads (Georgia DNR 2012). Opossums are considered a game animal and in many states there are rules and regulations pertaining to their harvest through trapping and hunting. Despite their appeal to hunters, biologists do not believe that hunting is a threat to most populations of this species (Georgia DNR 2012).

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APPENDIX B
ECOTOXICITY PROFILES

APPENDIX B
ECOTOXICITY PROFILES
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
AhR	aryl hydrocarbon receptor
AVS	acid-volatile sulfide
AWQC	ambient water quality criteria
BEHP	bis(2-ethylhexyl)phthalate
BERA	baseline ecological risk assessment
bw	body weight
CCC	Criterion Continuous Concentration
CMC	Criterion Maximum Concentration
$COPC_E$	chemical of potential ecological concern
CTR	critical tissue residue
dw	dry weight
EC_{50}	median effective concentration
EcoSSL	ecological soil screening level
ER-L	effects range-low
ER-M	effects range-median
EROD	ethoxyresorufin-*O*-deethylase
LC_{50}	median lethal concentration
LD_{50}	median lethal dose
LOAEC	lowest-observed-adverse-effect concentration
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed effect concentration
NOAEC	no-observed-adverse-effect concentration
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PeCDF	pentachlorodibenzofuran

RATL	Canadian Wildlife Service's Database of Reptile and Amphibian Toxicology Literature
SJRWP	San Jacinto River Waste Pits
SSD	species sensitivity distribution
TCDD	tetrachlorodibenzo-*p*-dioxin
TCDF	tetrachlorodibenzofuran
TCEQ	Texas Commission on Environmental Quality
TEF	toxic equivalent factor
TEQ	toxic equivalent
TEQ_P	toxic equivalent for PCBs
TRV	toxicity reference value
USEPA	U.S. Environmental Protection Agency
ww	wet weight

1 INTRODUCTION

This appendix provides summary ecotoxicity information for the chemicals of potential ecological concern (COPCEs) identified for the San Jacinto River Waste Pits (SJRWP) Site baseline ecological risk assessment (BERA). The profiles presented in this appendix provide the ecological effects evaluation performed for this BERA consistent with U.S. Environmental Protection Agency (USEPA) guidance (USEPA 1997). Each profile briefly describes the potential toxicity of each chemical or group of chemicals, addressing only those receptors for which the chemical is considered a COPCE. Each profile describes the types of potential toxic effects associated with exposure of terrestrial and aquatic biota to these chemicals, and provides the sources and supporting rationale for selection of individual toxicity reference values (TRVs). This section outlines the specifications for the scope of these toxicity profiles and methods for obtaining TRVs; subsequent sections provide the toxicity information needed for the BERA.

1.1 Receptor Surrogates

TRVs are identified for species considered representative of site-specific receptors or receptor surrogates. Site-specific receptor surrogates were selected to represent the following:

- Benthic macroinvertebrate communities
- Fish
  - Gulf killifish (*Fundulus grandis*)
  - Black drum (*Pogonias cromis*)
  - Southern flounder (*Paralichthys lethostigma*)
- Reptiles
  - Alligator snapping turtle (*Macrochelys temminckii*)
- Piscivorous birds
  - Neotropic cormorant (*Phalacrocorax brasilianus*)
  - Great blue heron (*Ardea herodias*)

- Invertivorous birds
  - Spotted sandpiper (*Actitis macularius*)
  - Killdeer (*Charadrius vociferous*)
- Semiaquatic mammals
  - Raccoon (*Procyon lotor*)
  - Marsh rice rat (*Oryzomys palustris*).

TRVs to address risks to these receptor groups are required for the BERA. Table B-1 provides a summary of the $COPC_{ES}$s for each ecological receptor group. Sections below address only those receptor–$COPC_{E}$ pairs shown in Table B-1.

1.2 Measures of Effect

Measurement endpoints and risk questions for the BERA are outlined in the RI/FS Work Plan (Anchor QEA and Integral 2010), and are discussed in the main text of the BERA. In summary, the following types of TRVs are needed for the BERA:

- Benthic macroinvertebrates
  - Bulk sediment concentration (mg/kg) for the benthic macroinvertebrate community
  - Concentrations in water (mg/L)
  - Critical tissue residue (CTR) values for dioxin and furan compounds (or other organics) expressed as concentration in whole clams (mg/kg wet weight [ww] or lipid)
- Fish
  - Concentrations in water (mg/L)
  - CTR values for dioxin and furan (or other organics) compounds expressed as concentrations in whole fish (mg/kg ww or lipid)
  - Concentrations of metals in media ingested by fish (mg/kg dry weight [dw])
- Reptiles, birds, and mammals
  - Daily ingested doses (mg/kg-day) for reptiles and mammals for all $COPC_{ES}$s, and for birds for $COPC_{ES}$s other than dioxins and furans

- CTR values for 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxin toxic equivalent (TEQ) concentrations in bird eggs (mg/kg ww).

As noted in the RI/FS Work Plan, the toxicity literature is often limited to studies reporting effects at the individual level. For this BERA, the types of individual effects measures are those clearly relating to population-level responses. These are generally survival, growth, and reproduction of tested individuals. Studies documenting an effect of a toxicant on an endpoint that is related by the authors of the study to survival, growth, or reproduction are also used (e.g., a developmental endpoint that is clearly related to the reduced survival of young). Studies addressing unrelated endpoints (e.g., cellular or biochemical alterations or gene expression) are generally not used to establish TRVs for the BERA, because these effects cannot be related to population-level assessment endpoints.

TRVs may be expressed as ingested doses, CTRs, concentrations in foods (fish only), or water concentrations, depending on the endpoint and receptor being evaluated. To calculate TRVs as an ingested dose, (i.e., mg/kg body weight [bw] per day) and where original toxicity studies report effect levels as concentrations in food of test animals but do not report body weight and/or consumption rate, values used by Sample et al. (1996) for body weight and consumption rate provided the basis for conversions to dose values, where needed. Table B-2 lists test species on which TRVs for birds and mammals were based and the default values for body weight and consumption rate for each. Dietary concentrations are presented in this appendix as dry weight, unless otherwise noted.

1.3 The Use of Uncertainty Factors in Estimates of Effects

The preferred approach for selecting TRVs is to find values that meet acceptability criteria (Section 1.4) and are taxonomically relevant and appropriate to the receptors of concern, but data may not be available for a given taxon or effect level of interest (e.g., a median lethal concentration [LC_{50}] may be available, but not a no-observed-adverse-effect level [NOAEL] or lowest-observed-adverse-effect [LOAEL]). In these cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV may be considered. In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty

factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk.

Chapman et al.'s (1999) review emphasizes the importance of evaluating the substance and context of the uncertainty. They caution against the extrapolation of LOAELs to NOAELs because there can be substantial uncertainty in moving from effects to no-effects concentrations. They provide several examples that support the use of uncertainty factors of 10 or less for individual extrapolations, including extrapolation of acute lethality toxicity tests to thresholds for sublethal effects in aquatic systems, and lowest-observed-effect concentration (LOEC) to no-observed effect concentration (NOEC) ratios for wildlife criteria (Chapman et al. 1999). This review points out that uncertainty factors are essentially screening tools for which the imprecision cannot be quantified, and should not be regarded as mathematical absolutes. These recommendations were used as a basis for the application of uncertainty factors in deriving TRVs where relevant effects level values were missing but related values were available.

1.4 Information Search Methods

These toxicity profiles draw from several well-established sources of TRVs and toxicity information commonly used at Superfund sites. Searches in the primary literature were also used to find information less readily available, to obtain the most recent information, and for chemicals for which it is necessary to evaluate toxicity in greater depth (e.g., dioxins and furans). The following describes the resources used to locate TRVs for the BERA.

1.4.1 Primary Literature and Compendia of Information for Superfund Sites

For bird and mammal TRVs for metals, this BERA draws largely from two widely accepted reviews of TRVs:

- The ecological soil screening levels (EcoSSLs) developed by USEPA (2005a)
- Sample et al. (1996).

Literature cited by USEPA in support of the EcoSSL values received careful and systematic scrutiny using specific and widely-used data quality criteria and with oversight by a large

panel of scientists (USEPA 2005g). Therefore, documents supporting the EcoSSL values were consulted as the preferred source for identifying the bird and mammal TRVs for metals.

Derivation of TRVs for aquatic life relied primarily on the following three references:

- Draft final BERA for Portland Harbor (Windward 2011a) (fish only)
- Long et al. (1995) (summarized in NOAA 1999; benthic macroinvertebrates only)
- USEPA ambient water quality criteria (AWQC), which are considered protective of 85 percent of aquatic species (Table B-3).

The draft final BERA for Portland Harbor in Portland, Oregon (Windward 2011a) presents a detailed review of the aquatic toxicological literature for many of the COPCs selected for fish at SJRWP. Their reviews include analysis of all published studies that could reasonably be found through literature searches, and selection of one or more values from the published data. The Portland Harbor BERA is in review by USEPA (draft final) and has had substantial USEPA input in its development. This compendium of information was used in this risk assessment to develop TRVs expressed as concentrations in fish foods. When well-established TRVs from Superfund sites, Sample et al. (1996), or the EcoSSL datasets are identified for use in this BERA, Integral did not conduct an independent evaluation of data quality using the original literature.

For benthic macroinvertebrates, a compilation of marine sediment benchmarks by Long et al. (1995) was used. Although other sources of marine sediment quality guidelines are available (MacDonald et al. 1996) and may be more robust on the basis of the methods used for their derivation, Long et al. (1995) is the same source of information used by the Texas Commission on Environmental Quality (TCEQ) in establishing sediment screening benchmarks for benthos. TCEQ interprets sediment chemistry in terms of risk to benthic invertebrate communities relative to Long et al.'s (1995) sediment benchmarks as follows:

- The effects range-low (ER-L) values are concentrations below which adverse effects on benthic communities rarely occur
- The effects range-median (ER-M) values are concentrations above which adverse effects on benthic macroinvertebrate communities are "probable"

- At concentrations between the ER-L and ER-M, adverse effects on benthic invertebrates are considered possible.

Although Long et al.'s (1995) ER-L and ER-M values have technical flaws (e.g., Sampson et al. 1996a, 1996b; Becker and Ginn 2008), they are regarded by TCEQ as protective of benthic communities. Therefore, in this risk assessment and consistent with the role of SQGs as screening benchmarks, ER-Ls were used to identify $COPC_Es$ and stations posing negligible risk to benthic macroinvertebrate communities. When concentrations of a $COPC_E$ in sediment exceed its respective ER-M value, the number of exceedances and area involved are considered to determine whether additional toxicity information is warranted to better describe risk.

When ER-L/ER-M values or other TRVs expressed as a bulk sediment concentration were not available for benthic macroinvertebrates, USEPA's AWQC for protection of aquatic life were used. AWQC are concentrations in water protective of 95 percent of species in an aquatic community, and are expressed as concentrations in water, as follows:

- The Criterion Maximum Concentration (CMC) is expected to be protective of aquatic life if the 1-hour average concentration in the waterbody does not exceed the CMC more than once every 3 years.
- The Criterion Continuous Concentration (CCC) is expected to be protective of aquatic life if the 4-day average concentration in the waterbody does not exceed the CMC more than once every 3 years.

If TRVs as sediment concentrations were not available to evaluate risk to benthic macroinvertebrate communities, a concentration in pore water was estimated using equilibrium partitioning models and was compared to the CCC from the available AWQC.

When AWQC and ER-L/ER-M values were not available to perform a screening comparison for fish and invertebrates, the following sources were consulted in descending order of preference:

- USEPA's ECOTOX database[1] (selected values were only from tests with marine species)
- Literature search for sediment or water toxicity data using the same databases listed above for toxicity data for birds and mammals.

When TRVs were not available from the sources listed above, a literature search for NOAEL and LOAEL values related to survival (or conversely mortality), growth, or reproductive endpoints was conducted using the following databases: Toxline, BIOSIS, Academic Search Complete, AGRICOLA, and GreenFile.

When a literature review was conducted, abstracts were reviewed to determine if an article reported on survival, growth, or reproductive endpoints for the relevant taxonomic group. Articles addressing these endpoints were retrieved for review and were evaluated according to the acceptability criteria described later in this section.

1.4.2 Toxicity Literature for Reptiles

The resources described above were consulted for information on the toxicity of COPCEs to reptiles. The availability of toxicity literature on reptiles is generally poor. One database of toxicity information has been developed specifically to house information on toxicity of chemicals to reptiles: The Canadian Wildlife Service's Database of Reptile and Amphibian Toxicology Literature (RATL) (Pauli et al. 2000). RATL is an annotated bibliography of literature related in some way to the exposure of reptiles to or toxicity of various chemicals, mostly metals, to reptiles and amphibians. It was searched for studies on reptiles that met the following criteria: laboratory study with exposure defined as environmental, dermal, oral, or injections, for all endpoints. Because RATL is now more than 10 years old, additional literature searches on the toxicity of COPCEs in reptiles were conducted.

There is a small body of literature that evaluates turtle tissue as a biomarker for organochlorine and metal contaminants (Beresford et al. 1981; Bergeron et al. 1994; de Solla et al. 1998; Keller et al. 2004; Robinson and Wells 1975). Most of the literature describes concentrations of chemicals in field-collected organisms but provides no means of

[1] http://www.epa.gov/med/Prods_Pubs/ecotox.htm

interpreting such information. The methods used to administer toxicants to turtles in most studies, such as painting of toxicants on to the surface of eggs, are not environmentally realistic, or have resulted in unrealistically high concentrations, and can be difficult to interpret for the purposes of risk assessment.

Very few toxicity studies addressing the COPCEs were located for reptiles in general, and none were found that presented daily ingested doses for comparison to exposure estimates for reptiles using the Site. None of the data found met the acceptability criteria for toxicity studies to be used in determining TRVs (below). The paucity of toxicity data for reptiles that is useful for risk assessment is confirmed by Weir et al. (2010) and Sparling et al. (2000a). These authors performed general literature reviews for ecotoxicological literature from 1978 to 1998 on vertebrates, and found that, of over 12,000 titles, only 163 papers address reptiles. The subject of most of these was turtles. About one-quarter of the studies address metals, a quarter address pesticides, 20 percent address polychlorinated biphenyls (PCBs), and the rest address "other" chemicals and effects of plastic and other debris.

Ecotoxicology of Amphibians and Reptiles (Sparling et al. 2000b) includes chapters on metals (Linder and Grillitsch 2000) and organic chemicals other than pesticides (Portelli and Bishop 2000). For both categories, these investigators found that the majority of information is from uncontrolled studies using field-collected organisms in which exposure to several xenobiotics could have occurred or are documented to have occurred. Most studies report tissue concentrations, not dose-response information. According to Linder and Grillitsch (2000, p. 398), "[t]here is a collective agreement that for reptiles little to no explicit information on the toxicological effect potential is available for any metal." Similarly, the majority of publications report dioxin, furan, and PCB concentrations in tissues of reptiles collected in the field. Bis(2-ethylhexyl)phthalate (BEHP) is not mentioned in the book (Sparling et al. 2000b).

In their review, Portelli and Bishop (2000) found no reports of reptiles dying as a result of PCB, dioxin, or furan exposure, despite fairly elevated concentrations in their tissues. Bishop et al. (1991) reported developmental abnormalities (e.g., abnormal eyes, claws, and bills) and behavioral abnormalities in turtles exposed to dioxins, furans, PCBs, and organochlorine pesticides, but dose-response relationships have not been reported. This and other studies

cited by Portelli and Bishop (2000) suggest correlations between concentrations of PCBs, polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) and abnormalities in developing embryos, but these data are confounded by the presence of pesticides and other chemicals in the environment and tissues of organisms studied. Portelli and Bishop (2000) note that there is no correlation between dioxin, furan, and PCBs in eggs and incidence of abnormalities when TEQ was used to characterize exposure, regardless of the toxic equivalent factor (TEF) scheme used. The available information on toxicity of dioxins and dioxin-like compounds to reptiles is reviewed in Section 2.

Because of the lack of information on the toxicity of COPCEs to reptiles, TRVs were not developed for reptiles, and risks to reptiles at the Site cannot be evaluated using available methods and data. Exposure assessment is conducted, and risks to reptiles are discussed in the uncertainty analysis.

1.5 Acceptability Criteria

The toxicity literature reflects a wide range of investigator objectives, most of which were not associated with ecological risk assessment. As a result, the technical quality of toxicological studies potentially available for risk assessment varies widely. Some of the available literature is not acceptable for use in a BERA. Because most studies, especially older research, provide imperfect ecotoxicological information, guidelines to evaluate the acceptability of literature used to derive TRVs are needed. The use of basic standards for data quality ensures that the meaning and uses of the reported information are clear. The following are among the most important considerations for inclusion of toxicity data in the BERA:

- Methods must be clearly presented and complete.
- The test subjects should not have been exposed to toxicants other than the toxicant under study prior to or during the investigation, unless the pre-existing exposure is addressed by the study. For field studies in which test subjects have been exposed to other chemicals, NOAELs can be derived.
- Exposures to toxicants in water are not performed with solutions within which the toxicant concentration exceeds its water solubility.
- Either an effects level (e.g., LOAEL) or a no-effects level (e.g., NOAEL) is reported.

- Investigators use and report results for an experimental control, and control media are identical to exposure media in every way except for the toxicant under study.
- The statistical design employs an appropriate number of replicates, treatments are randomized, and the level of significance is reported for differences in response from controls.
- The tested endpoint is clearly related to the survival, growth, or reproduction of the tested subjects.
- There are no obvious confounding factors, such as limited feeding of tested specimens, which could affect the test endpoint.

To the extent that a study that is selected to support the risk evaluation deviates from these guidelines, the uncertainties associated with the TRV, and therefore the risk evaluation, tend to increase.

In addition to the above guidelines, preference is given to toxicity studies with the following characteristics:

- Both a LOAEL and a NOAEL are reported.
- The form of the test chemical is reported, and is a form commonly found in the environment.
- Tissue residue-based TRVs report concentrations for whole-body samples (because concentrations of individual organs and isolated tissues such as liver or gill tissue of receptors at the Site cannot be reliably predicted and were not measured) or eggs.
- Concentrations in exposure media or tissue are measured, not estimated.
- Exposure duration is clearly reported, and effects of chronic exposures are evaluated.
- A standard or peer-reviewed study protocol is used.

Where high quality TRVs are not available, a conservative approach to developing points of comparison for exposure estimates is used. Estimated exposures falling below any of the TRVs selected indicate a conclusion of no risk with a high degree of confidence. If one or more studies generally meeting most of these acceptability criteria could not be identified for a given measurement endpoint, and estimated Site-specific exposures exceed the TRV, risks were described qualitatively and were discussed in the uncertainty section of the risk assessment report.

1.6 Methods Used for Aggregation of Toxicity Data

For most COPCEs, reasonably conservative TRVs from the literature are compared directly to Site-specific exposure estimates. In these cases, the TRV reflects results of a study or studies of acceptable quality to provide the best representation of the receptor on the basis of taxonomy and sensitive life stages of the Site-specific receptor. If the estimated exposure is less than the individual TRV, no further analysis is conducted, and risk is considered negligible if both the central tendency exposure and reasonable maximum exposure are below the LOAEL ($HQ_L < 1$).

For dioxins, furans, and PCBs, there is a large body of literature describing toxicity to various species. Risks due to both 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) TEQs and total PCBs are considered. For these COPCEs, several TRVs of equal quality and relevance were available in some cases. If fewer than 10 values were found, the following steps were taken to derive a TRV:

1. Within-species NOAELs or LOAELs were grouped.
2. The geometric means of the within-species NOAELs and the geometric mean of the within-species LOAELs were calculated.
3. Resulting geometric means within a TRV category (LOAEL or NOAEL) are pooled. No individual species is represented by more than one value, although some values are the results of only one study.
4. The geometric mean of the pool of data for multiple species is calculated, and that value becomes the NOAEL or the LOAEL for the COPCE and receptor.

The RI/FS Work Plan indicates that cumulative distribution functions derived from multiple effects-level metrics with a species, or species sensitivity distributions (SSDs), would be developed using multiple literature values for several species. This is a tool that can be used to clearly define the risk and the uncertainty associated with a risk calculation. However, sufficient data for a set of related taxa that have similar exposure and effects metrics were not found, except for the SSD for early life stage fish developed for Steevens et al. (2005).

A method to extrapolate TRVs between species on the basis of the difference in body weights between the two species, called allometric scaling, has been used at some Superfund sites. The technical basis for extrapolation of TRVs between species based on body size is not as well established for ecological receptors as it is for extrapolations relating to human health risk assessment (i.e., rat to human extrapolations), where it is most widely applied. Because of uncertainty in the use of allometric models to scale TRVs between species, particularly for birds, extrapolations on the basis of body size was not used to estimate or derive measures of effects when species-specific TRVs are not available.

2 ORGANIC COMPOUNDS

The organic chemicals considered COPCEs for one or more ecological receptors are dioxins, furans, PCBs, BEHP, carbazole, and phenol (Table B-1). Because dioxins and furans are the indicator chemical group at the Site, and because their toxicity is relatively well studied, greater depth of information is provided for them than for other organic COPCEs. PCBs are also discussed at length because their toxicity in vertebrates can be considered additive with the toxicity of dioxins and furans, and the BERA evaluates risk using this additivity model.

2.1 Dioxins and Furans

Attachment B2 to the Screening Level Ecological Risk Assessment (SLERA) (Appendix B in Anchor QEA and Integral 2010) provides an overview of the technical literature available for the evaluation of dioxin and furan toxicity to birds, mammals, fish, reptiles, and invertebrates. The toxicity profile presented below repeats much of that information (e.g., for benthic macroinvertebrates) and provides a more focused summary and rationale for selection of studies that provide the basis for TRVs or SSDs to be used.

The main text of the BERA (Section 3.2) also repeats some of the basic information from Appendix B, Attachment B2 to the RI/FS Work Plan on the general toxicology of dioxins and furans, and the common conceptual framework used to present and evaluate dioxin and furan toxicity information for vertebrates. Readers are referred to that text for discussion of the basis for the use of TEFs and 2,3,7,8-TCDD TEQ concentrations or doses for assessment of exposure and toxicity to dioxins, furans, and dioxin-like PCBs. The approach used in this BERA is consistent with USEPA (2008) guidance.

From the perspective of an assessment of ecological risks, adverse effects of dioxins and furans on reproductive success, growth, and survival are relevant to evaluating the potential for population-level effects in any receptor. A range of reproductive and developmental effects such as reduced fertility, early-stage embryotoxicity, early life-stage mortality, developmental effects, and reduced growth of offspring are relevant, because these effects can conceivably affect the growth or viability of a population. This section provides a summary of information considered in the development of dioxin and furan TRVs for

ecological receptors addressed by this BERA, identifies the TRVs to be used in the risk evaluation, and provides supporting rationale for their selection.

2.1.1 Benthic Macroinvertebrates

Several studies have found no adverse effects in aquatic invertebrates following exposure to TCDD; studies to provide systematic toxicity data for the other dioxin and furan congeners are rare. Data summarized in Attachment B2 to the SLERA include findings of no effects for the following:

- **Estuarine amphipods.** Barber et al. (1998) exposed the estuarine amphipod *Ampelisca abdita* to sediments spiked with 2,3,7,8-TCDD at concentrations between 1.1 and 25,000 ng/kg dw. No significant differences ($p > 0.05$) were found for survival or growth between any of the spiked sediments and the negative control sediments. Barber et al. (1998) identified an NOEC for 2,3,7,8-TCDD in sediment of 25,000 ng/kg for the amphipod.
- **Freshwater insect and oligochaete.** West et al. (1997) exposed the freshwater chironomid (*Chironomus dilutus*) and oligochaete (*Lumbriculus variegatus*) to nominal dietary concentrations of 30, 300, and 3,000 ng 2,3,7,8-TCDD/kg of total organic carbon (TOC) in food over 28-day (*Lumbriculus*) and 35-day *(Chironomus*) exposure periods. Maximum 2,3,7,8-TCDD body burdens in *Chironomus* and *Lumbriculus* were 5,084 and 9,533 µg/kg lipid at the 3,000 ng/kg TOC[2] treatment levels. No significant effects were found on survival, growth, or reproduction for either of the test species. Another study in which chironomids (*Chironomus riparius*) were exposed to 2,3,7,8-TCDD in spiked sediments with concentrations ranging from <150 to 10,000 ng/kg dw reported no significant effects on survival or growth and no significant differences in the occurrence of deformities from control (Loonen et al. 1996). The maximum observed tissue concentration was 14,000 ng/kg dw.
- **Marine polychaetes, musssels, and grass shrimp.** Pruell et al. (1993) and Rubenstein et al. (1990) evaluated the toxicity of 2,3,7,8-TCDD and 2,3,7,8-tetrachlorodibenzofuran (TCDF) on polychaetes (*Nereis virens*), bivalve molluscs (*Macoma nasuta*), and grass shrimp (*Palaemonetes pugio*) exposed to sediments collected from the Passaic River, New Jersey. The mean sediment concentrations of the two compounds were 656 and

[2] Measured concentrations were 3,804 ng/g TOC for *Chironomus* diet and 3,594 ng/g TOC for *Lumbriculus* diet.

334 ng/kg dw, respectively. The final tissue concentrations for the polychaetes, bivalves, and shrimp were 422, 142, and 138 ng/kg, respectively. There were no major differences in survival between the test and reference-area sediments, with control-adjusted survival of all three species being greater than 90 percent.

- **Freshwater zooplankton and snails.** Adams et al. (1986) evaluated the toxicity of 2,3,7,8-TCDD to daphnids (*Daphnia magna*) in 48-hour water exposures at concentrations of 0.2 to 1,030 ng/L, followed by a recovery period. The authors concluded that no toxic effects were found. Yockim et al. (1978) evaluated the toxicity of 2,3,7,8-TCDD to daphnids (*Daphnia magna*) and snails (*Helisoma* sp.) in 32-day water exposures at concentrations of 2.4 to 4.2 ng/L. The authors found no adverse effects on growth, reproduction, or feeding for either test species. Isensee and Jones (1975) evaluated the toxicity of 2,3,7,8-TCDD to snails (*Physa* sp.) and daphnids (*Daphnia magna*) in water exposures at concentrations of 0.05 to 1,300 ng/L. The authors concluded that no effects on reproductive activity, feeding, or growth were found for either test species.

One study reported adverse effects on freshwater crayfish (*Pacifastacus leniusculus*), a crustacean, following injections of 2,3,7,8-TCDD, but the results are not considered useful for risk assessment because of several uncertainties (Ashley et al. 1996). Although mortality was observed, the authors concluded that the cause of mortality was not associated with tissue pathology, and could not specify the cause of the observed mortality. In addition, the authors acknowledged that their sample sizes were small, with only three to six crayfish exposed to each 2,3,7,8-TCDD concentration in three separate experiments. Finally, the methods indicate that excessive dimethylsulphoxide may have been used in dosing solutions, which could have contributed to mortality.

The available published studies on this topic are summarized in Attachment B2 to the SLERA (Anchor QEA and Integral 2010), included here as Table B-4. This compilation of literature and related analysis finds that, in contrast to fish and wildlife, most studies of aquatic invertebrates have found that invertebrates are relatively insensitive to TCDD toxicity. Although aryl hydrocarbon receptor (AhR) homologues have been identified in various invertebrate species, invertebrate AhR homologues lack the ability to bind dioxins (Hahn et al. 1992; Butler et al. 2001). However, recent studies have documented reproductive toxicity

of 2,3,7,8-TCDD in bivalve molluscs. The mechanism by which dioxins affect bivalve molluscs has yet to be identified with certainty, but researchers agree that it is independent of AhR homologues.

The only series of studies of acceptable quality (Section 1.4) showing effects of 2,3,7,8-TCDD on invertebrates involve injection of TCDD into the eastern oyster (*Crassostrea virginica*) and the soft-shell clam (*Mya arenaria*), both bivalve molluscs. They found that 2,3,7,8-TCDD preferentially accumulates in the gonads and digestive glands of the bivalves, which is consistent with the earlier findings of Rhodes et al. (1997). They speculated that this uptake was not solely related to lipid content, but was "best explained by 2,3,7,8-TCDD binding to a tissue-specific receptor" (Wintermyer et al. 2005).

Cooper and Wintermyer (2009) found a time-dependent loss in body mass and microscopic abnormalities in multiple tissues in clams following a single exposure administered by gavage of 200 ng 2,3,7,8-TCDD/kg tissue ww, but no loss in body mass following 24-hour waterborne exposure or muscle injection to achieve the same tissue concentration. Oysters exhibited decreases in gonadal development in females at tissue concentrations of 2.0 ng/kg, which is consistent with the sensitivity found by Wintermyer and Cooper (2003, 2007). Although gonadal development in the clams was not evaluated by Cooper and Wintermyer (2009), Butler et al. (2004) found a lack of proper gonadal development in both female and male clams at comparable 2,3,7,8-TCDD concentrations. Wintermyer and Cooper (2007) found a shift in the male/female ratios for the oysters, with a decrease in the number of females, at tissue concentrations of 10 ng/kg (Table B-4).

Although the use of injection and gavage may not precisely mimic processes of exposure and uptake in environmental settings, these recent studies provide useful information, updating the literature on the toxicity of 2,3,7,8-TCDD to some invertebrates. Cooper and Wintermyer (2009) conclude that their data, together with the studies they reviewed, provide evidence for sensitivity of reproductive endpoints in bivalve molluscs to 2,3,7,8-TCDD exposure, and that tissue concentrations that resulted in altered gonadal development and reduced larval survival in the laboratory (i.e., 2 to 10 ng/kg) were comparable to the levels observed in field populations of *M. arenaria* (4.8 to 20 ng/kg) and *C. virginica* (0.15 to 3.2 ng/kg) in chemically contaminated waterbodies in New Jersey (i.e., Newark Bay, Arthur

Kill, and Sandy Hook), where bivalves are known to be stressed. Cooper and Wintermyer (2009) concluded that 2,3,7,8-TCDD alters normal development of reproductive organs and larval development in tested bivalves at whole-organism tissue concentrations of 2 to 20 ng/kg, although they acknowledge that the estuaries they evaluated were affected by numerous chemicals other than 2,3,7,8-TCDD.

However, Cooper and Wintermyer (2009) draw the conclusion, that survival of oyster larvae is impaired by TCDD at 2 ng/kg tissue from their 2003 field study (Wintermeyer and Cooper 2003). This may overstate the role of TCDD in survival of larvae. Wintermyer and Cooper (2003) transplanted wild-caught adult eastern oysters (*Crassostrea virginica*) to Newark Bay, the Arthur Kill area of Raritan Bay, and Sandy Hook, New Jersey. Results suggest that oysters with TCDD (ng/kg)/TCDF (ng/kg)/total PCB (µg/kg) concentrations of 3.2/2.1/68 and of 1.3/1.7/65 had reduced survival of veliger larvae. Conditions of this study are not analogous to conditions at the SJRWP site because of the relatively high levels of PCBs in the oyster tissue, which could have been the cause of reductions in larval survival, found in combination with the TCDD levels that are reported. Also, Wintermyer and Cooper (2003) exposed test organisms in complex urban estuaries, where sediment and water quality are influenced by oil refineries, urban runoff, combined sewer overflows, sewage treatment plants, and other sources of anthropogenic pollutants. The effects of estrogenic compounds and other chemicals in addition to TCDD, TCDF, and PCBs were not considered or discussed by Wintermyer and Cooper (2003), and exposures of test organisms to other chemicals were not evaluated. However, Wintermyer and Cooper (2003) also exposed oysters to TCDD alone in a controlled experiment, and found reduced larval survival at the lower tissue concentration (2 ng/kg ww). Therefore, although the field study cannot account for the effects of the mixtures, the laboratory study demonstrates that 2 ng/kg ww in whole eastern oyster tissue causes reduced fertilization and reduced larval survival in eastern oysters.

According to the earlier analysis of this information (Attachment B2 to the SLERA), the assessment of risks associated with dioxin exposures to molluscs will be based on the assumption that bivalve molluscs are among the most sensitive invertebrate taxa, and that evaluations based on toxicity of TCDD to bivalves are protective of benthic macroinvertebrates as a group. Therefore, the TRVs for performing the risk evaluation for the benthic macroinvertebrate community and for bivalves are as follows:

- **Benthic macroinvertebrate community.** A no-observed-adverse-effect concentration (NOAEC) only, calculated as the geometric mean of survival NOAECs reported in spiked sediment bioassays summarized in Table B-4, will be used in the BERA. The geometric mean of these NOAEC values is 2,343 ng/kg dw (Barber et al. 1998; Pruell et al. 1993; Rubenstein et al. 1990; Loonen et al. 1996).
- **Bivalves.** The studies summarized by Cooper and Wintermyer (2009) indicate that tissue concentrations in the range of 2 to 20 ng/kg may cause adverse reproductive and developmental effects in bivalve molluscs (Table B-4). The lowest-observed-adverse-effect concentration (LOAEC) of 2 ng TCDD/kg ww tissue for delayed gonadogenesis in males and histopathology in females (Wintermyer and Cooper 2007) and reduced egg fertilization and larval survival (Wintermyer and Cooper 2003) was chosen for use in the BERA. A corresponding NOAEC was not available.

Although Table B-4 presents NOAECs and LOAECs using a variety of metrics, the approach for the benthic macroinvertebrate community was selected because sediment concentrations have been empirically measured at the Site and in the supporting studies, eliminating any need for modeling. Also, the studies supporting derivation of the NOAEC above span several major taxonomic groups, including arthropods, crustaceans, annelids, and molluscs. In light of the absence of effects on most other invertebrate taxa at tissue concentrations greater than the effects levels reported by Cooper and Wintermyer for clams and oysters, the use of bivalve molluscs as a surrogate benthic invertebrate taxon for evaluating the exposure and potential effects of TCDD on benthic macroinvertebrates generally at the Site would be an overly conservative means to address risk to benthic macroinvertebrates as a group. Results of comparisons of concentrations in clam tissue to the CTRs from studies with clams and oysters are therefore considered applicable only to assessment of risks to bivalves.

2.1.2 Fish

Some fish species appear to be among the most sensitive of vertebrates to dioxin and furan toxicity and are thought to be the most sensitive of aquatic taxa (USEPA 2008). Dioxin toxicity in fish is mediated via the AhR pathway, as it is in birds and mammals. Dioxins, individual 2,3,7,8-substituted congeners, and mixtures of dioxin-like compounds produce similar early life-stage toxic effects in fish, supporting the conclusion that toxicity is

mediated through a common mechanism (Walker et al. 1996). Unlike mammals, which possess a single form of AhR, fish can have multiple AhR homologues, possibly due to a gene duplication event that occurred during the evolution of fish species (Carney et al. 2006; Andreasen et al. 2007). It is not yet clear what role the different homologues play in dioxin toxicity to fish (USEPA 2008), but this information suggests the possibility of substantial variation in sensitivity among fish species.

As for other ecological receptors, early life stages are the period of greatest sensitivity of fish to dioxin toxicity (Walker and Peterson 1991; Elonen et al. 1998; Steevens et al. 2005; Carney et al. 2006). Multiple fish species, including brook trout, catfish, northern pike, fathead minnow, zebrafish, and medaka, have been shown to be particularly sensitive to dioxin toxicity during the life stages from hatching to swim-up (following absorption of the yolk sac and transition into the water column for feeding) (Walker and Peterson 1991; Elonen et al. 1998). Toxic effects to the egg are seen during later embryonic development as well. For example, embryonic zebrafish exposed to 2,3,7,8-TCDD within a few days post-fertilization begin to manifest toxic responses, including pericardial and yolk sac edema (described further below), reduced cardiac function, and alterations to cartilage growth at 48 to 120 hours post-fertilization, when morphogenesis of primary organ systems and embryo growth are occurring (Carney et al. 2006).

Studies exposing post-swim-up trout fry to concentrations of dioxins associated with significant increases in mortality in the pre-swim-up fry did not find significant mortality in the later life stages (Walker and Peterson 1991; Walker et al. 1996). Possibly because the importance of early life stage toxicity in fish was established relatively early, sublethal effects of dioxins and furans on juvenile and adult fish, including potential effects on feeding, growth, predator avoidance, and other functions important to fish survival and reproduction, are not as well studied (Carney et al. 2006). It is also notable that the literature suggests that population resistance to dioxin toxicity can also occur over time in some fish, as shown for a killifish population living in the vicinity of a Superfund site with high dioxin levels (Nacci et al. 2002).

2.1.2.1 Reproductive Effects

Effects on reproduction, including decreased egg productivity (number of eggs produced per female) and decreased spawning success (production of eggs that are successfully fertilized) have been observed in fish following chronic exposures in experiments with dioxins that result in tissue concentrations in the range of nanograms per gram. No effects on fertility were found in adult brook trout exposed for 28 days to a range of dioxin concentrations targeted to achieve 0, 75, 150, 300, 600, and 1,200 ng/kg adult tissue (which achieved a concentration range of up to 517 ng/kg egg tissue through maternal transfer) (Johnson et al. 1998; Tietge et al. 1998). However, chronic dietary exposure of adult female zebrafish to an estimated dose of 0, 80, 320, or 800 pg TCDD/day for 20 days (corresponding to measured body burdens of 0, 1,100, 6,900, and 15,000 ng/kg at the end of the exposure period) led to significant adverse reproductive effects in the two highest exposure groups, including decreased egg production. At the highest exposure, a reduction in the number of ovarian follicles and decreased spawning success of up to 80 percent relative to control were observed (Heiden et al. 2009).

2.1.2.2 Developmental Effects

Developmental effects are manifested in early life stages during critical developmental processes in embryonic and newly hatched fish. They are often symptomatically similar to "blue sac disease," a disease of yolk sac fry that was first characterized as a response to poor conditions in hatcheries. Blue sac disease is characterized by edema, or liquid accumulation, in the yolk sac, causing swelling, which can lead to reduction or destruction of circulatory function in the yolk sac and/or body. Experimental exposure of fish eggs to 2,3,7,8-TCDD has led to increased incidence of symptoms very similar to blue sac disease, including subcutaneous edema with loss or cessation of blood circulation in the yolk sac and body (Spitsbergen et al. 1991; Carney et al. 2006).

Additional effects of dioxin exposure at the egg stage that manifest after hatching and prior to swim-up can include necrosis of the retina, brain, and spinal cord, malformations of the tail fin, microcephaly, and deformities of mandibular and opercular bones (Spitsbergen et al. 1991; Elonen et al. 1998; Johnson et al. 1998). Johnson et al. (1998) noted symptoms of edema in brook trout fry at lower exposure concentrations (84 ng/kg egg) than

concentrations associated with significant incidence of opercular and mandibular deformities (156 ng/kg).

Cardiovascular dysfunction is regarded as an important adverse effect of 2,3,7,8-TCDD in fish, and is particularly well studied in zebrafish, which are frequently used as an animal model for 2,3,7,8-TCDD toxicity (Carney et al. 2006). In addition to pericardial edema, toxic effects of 2,3,7,8-TCDD on cardiovascular development in fish include inhibition of growth and normal development of the common cardinal vein, a paired vessel that grows across the yolk, connects to the heart, and is extensively reorganized during later embryonic development (Bello et al. 2004). 2,3,7,8-TCDD exposure was observed to cause physiological alteration in atrioventricular and bulboventricular valve development of the zebrafish, leading to an inability of the heart to function effectively in circulating blood (Mehta et al. 2008). Although zebrafish have been the most intensively studied species with respect to mechanisms of dioxin effects on cardiotoxicity, cardiac effects of dioxin exposure, including pericardial hemorrhage and myocyte necrosis in trout fry exposed to 2,3,7,8-TCDD have been shown for other fish species as well (Spitsbergen et al. 1991). Linking cardiovascular effects in fish to ecological endpoints in the BERA is not straightforward and is likely prohibitively uncertain in a BERA context. Nevertheless, the available information on cardiovascular effects is useful in understanding the ways dioxins may affect wild fish at the Site.

2.1.2.3 Effects on Growth

Sublethal effects of 2,3,7,8-TCDD exposure can include malformations of cartilage in the developing fish, leading to reductions in length of or alterations to parts of the skeletal structure (Spitsbergen et al. 1991; Carney et al. 2006). Growth, as measured by body length or weight in juveniles, was reduced in white sucker exposed at the egg stage to 2,3,7,8-TCDD resulting in a tissue concentration of 1,220 ng 2,3,7,8-TCDD/kg egg tissue and in lake herring at 717 ng/kg egg tissue (Elonen et al. 1998). Elonen et al. (1998) further suggested that reduced lengths observed in fish exposed to 2,3,7,8-TCDD were related to the manifestation of edema, leading to prevention of blood flow through the yolk sac vasculature, and ultimately resulting in decreased absorption of nutrients to the body.

2.1.2.4 Development of a Species Sensitivity Distribution for Fish

Among tested freshwater fish species, sensitivity to 2,3,7,8-TCDD-induced early life stage toxicity ranges approximately 50-fold, with salmonids being the most sensitive and zebrafish the least sensitive species (Walker and Peterson 1991; Elonen et al. 1998; USEPA 2008). Steevens et al. (2005) compiled data on effects of dioxins and furans to embryos from 10 studies of several fish species, generating a summary of the geometric means of NOAEC and LOAEC sublethal growth and reproduction endpoints ranging from 0.42 µg/kg lipid for lake trout to 60.28 µg/kg lipid for zebrafish. They also compiled lethal effects (LR_{50}) concentrations ranging from 0.53 µg/kg lipid for lake trout to 153.53 µg/kg lipid for zebrafish. The similarity in the ranges of the sublethal and lethal effect concentrations reflects the steep dose-response associated with dioxin toxicity in fish (i.e., the transition from a concentration that causes an observable sublethal effect to a concentration causing a lethal effect occurs over a small range). For some salmonids (e.g., brook trout and lake trout), this transition occurs within less than a 1 ng/g increase in concentration (Elonen et al. 1998; Steevens et al. 2005).

Steevens et al. (2005) fitted the fish egg tissue residue data to a logistic distribution to generate an SSD based on the geometric mean of the LOAEC and NOAEC for all 10 studies, and a second one using the LC_{50} values (i.e., concentrations lethal to half the test organisms). The 10 geometric means (Table B-5) were used to generate an SSD for fish exposed to TCDD and dioxin-like compounds. Using the resulting SSD, Steevens et al. (2005) generated egg tissue residue-based TRVs for dioxin-like compounds in fish tissue that are protective of specified percentiles (e.g., 95, 97.5, and 99 percent) of species, with confidence limits. The SSD developed by Steevens et al. (2005) is used to evaluate effects of dioxin and furan exposures in fish for this BERA. By necessity, this risk assessment uses TEQ_F concentrations in whole body samples of fish for comparison to the CTRs of Steevens et al. (2005). This approach conservative. Tietge et al. (1998) found that TCDD concentrations in eggs of brook trout (*Salvelinus fontinalis*) were just 39 percent of the concentrations in the whole fish. Heiden et al (2005) reported an even lower level of egg accumulation of TCDD relative to female whole bodies in zebrafish, with egg concentrations of just 5 percent of whole adults. This risk assessment is conservative because it assumes a 1 to 1 ratio of whole adult fish to egg concentrations.

Steevens et al. (2005) remark that because so many of the species represented in the SSD are salmonids, which are generally very sensitive to many toxicants, the resulting toxicity residue benchmarks derived from the SSD are conservative for many non-salmonid fish species. There are no fish receptor surrogates for the SJRWP that are salmonids, so the SSD derived by Steevens et al. (2005) is considered to be conservative for application at the Site.

2.1.3 Reptiles

Limited data are available regarding the toxicity of dioxin-like compounds to turtles, and the potential effects of dioxins and furans on other reptiles have not been studied (Portelli and Bishop 2000). Studies with turtles have generally been conducted in the field, where exposures to other chemicals may occur, so conclusions about the effects solely from exposure to dioxins and furans are not possible. The available data for turtles suggest that additional controlled studies of adverse effects are needed to understand dioxin and furan toxicity in this taxon. The available studies are summarized below.

2.1.3.1 Ethoxyresorufin-O-Deethylase Induction in Snakes

Liver cells from the African brown snake (*Lamprophis fuliginosus*) were exposed *in vitro* to TCDD and four non-ortho substituted co-planar (i.e., dioxin-like) PCB congeners (PCB77, PCB81, PCB126 and PCB169) (Hecker et al. 2006). Dose-dependent increases in ethoxyresorufin-O-deethylase (EROD) activity were observed with exposure to TCDD and PCB126, but not with the other PCB congeners, suggesting lower sensitivity of this snake than other vertebrates to the dioxin-like toxicity of PCBs. The potency in EROD induction by PCB126 relative to TCDD was comparable to potency in mammals and the more sensitive birds, but indicated a higher sensitivity of the snakes than of fish. This information cannot be used to interpret estimated exposures in the field.

2.1.3.2 Reproductive Effects

Bishop et al. (1991) documented an increase in unhatched eggs and deformities in snapping turtles collected from an area contaminated with multiple potential toxicants, including PCBs, dioxins, and furans. Because the exposure was to a mixture, it is not possible to attribute the effects to one or a specific subgroup of the chemicals measured.

In a separate study (Bishop et al. 1998), turtle eggs exposed to a mixture of dioxins, furans, and PCBs (as well as pesticides and other chemicals) collected from the field had significantly increased proportion of unhatched eggs and increased proportion of deformed hatchlings. However, data from this study are limited in their usefulness, because of the presence of multiple contaminants and because correlation between individual contaminants and adverse outcomes was not conducted. Portelli and Bishop (2000) indicate that the rates of abnormalities observed in these two studies correlated with dioxin, furan, and PCB concentrations in eggs, but not when they were expressed as TEQ.

More recently, a field study and a controlled experiment addressing the potential for hormonal alterations and reproductive effects in turtles have been conducted. Evaluation of plasma hormone levels in male yellow-blotched map turtles collected from a TCDD-contaminated area revealed that estradiol increased and testosterone decreased in a small proportion of turtles (Shelby and Mendonça 2001). The potential reproductive consequences of the observed changes in hormone levels are not clear. Although some evidence of turtle sex reversal has been observed following exposures of turtles to PCBs (Willingham et al. 2000), raising concerns that this might be a result of dioxin-like toxicity, 2,3,7,8-TCDD has been shown not to cause sex reversal following administration to eggs in a laboratory study (Gale et al. 2002). Sexual development of reptiles is also linked to incubation temperature (e.g., Nichols et al. 1999), in the absence of chemical contamination, suggesting that toxicological endpoints such as those listed above be interpreted with caution.

Quantitative measures of effects are not available for evaluation of the risks of exposures of reptiles to dioxins and furans at the Site. A quantitative estimate of ingestion exposure to snapping turtles is planned, and this can be used to evaluate the reptile exposures relative to those of other receptors. In the absence of new information by the time the BERA is drafted, risks to reptiles will be evaluated qualitatively.

2.1.4 Birds

Evaluation of the toxic effects of dioxins and furans has been conducted in multiple North American bird species, including herons, egrets, terns, cormorants, a bluebird species, chickens, pheasants, and ducks. Toxicity of dioxins and furans in birds is mediated through

AhR, and may also occur through other biochemical pathways. Endpoints evaluated in birds have included reproductive success (using a variety of endpoints), thyroid toxicity, cardiovascular toxicity, immune toxicity, and effects on growth and survival.

As for fish, AhR-mediated toxicity is the focus of this toxicity profile for birds because AhR-mediated effects are assumed to occur at lower doses than other effects. Also, effects on early life stages, which are well documented for birds exposed to dioxins, furans and dioxin-like PCBs (e.g., Henshel et al. 1997), are also emphasized because they are relevant for understanding risks to bird populations.

As described in Appendix B, Attachment B2 to the RI/FS Work Plan (Anchor QEA and Integral 2010), two lines of evidence are used in the BERA to evaluate risks to birds from exposures to dioxins and furans: comparison of estimated daily ingested doses to TRVs expressed in the same terms (mg/kg bw-day), and comparison of estimated egg concentrations to TRVs expressed in the same terms (ng TEQ/kg egg ww). This section does not repeat the information presented in Attachment B2; instead it expands on that discussion, focusing on data supporting development of TRVs for these two lines of evidence.

2.1.4.1 Variability in Avian Toxicity of Dioxins and Furans

Exposure to dioxin-like compounds is associated with both embryo mortality and a variety of adverse effects on chick development, including reduced embryo and hatchling growth, deformities, and abnormalities in the developing heart. The exposure levels at which adverse effects are observed span a large range across avian taxa, from low nanograms per kilogram to low micrograms per kilogram in tissue.

There are clear differences among bird species in susceptibility to dioxin-like toxicity, which have been attributed to biochemical differences in AhRs among species (Karchner et al. 2006; Head et al. 2008). Domestic chickens are generally considered to be the most sensitive bird species tested, not only in responses to TCDD exposure, but also in responses to other dioxin-like compounds, such as TCDF and PCB126 (TN & Associates 2002). Even the sensitivity among chicken species measured as EROD induction (by PCB 126) varies by a factor of 20, with chicken species having greater sensitivity than species with wild populations. The next

most sensitive species (as measured by EROD induction) is the pheasant (TN & Associates 2002), followed by turkeys, ducks, and gulls. Using a different set of assays, common terns were significantly less sensitive to TCDD toxicity than chickens. Further, according to TN & Associates (2002), an experiment by Sanderson et al. (1998) compared EROD induction in bird hepatocyte cultures for several species. Among other things, this study reports 40-fold and 80-fold variation in EC_{50}s for EROD induction in TCDD-exposed hepatocytes of ring-billed gulls and double crested cormorants, respectively. Although intraspecies sensitivity is not often discussed, these studies demonstrate that species-specific differences are relevant to understanding ecological risks.

The general finding that chickens are most sensitive has been verified for EROD induction and egg mortality, but is less clear for embryo developmental endpoints. Cohen-Barnhouse et al. (2011) found that changes in developmental endpoints in embryos of chickens, pheasants, and quail were not consistently related to dose of TCDD, TCDF, or 2,3,4,7,8-pentachloro-dibenzofuran (PeCDF). Although their results generally supported the view that chickens are the most sensitive for egg mortality, developmental abnormalities occur at different stages following egg laying, and these effects did not follow simple dose-response relationships. Cohen-Barnhouse et al. (2011) report that post-hatch mortality of surviving chicks under all treatments did not differ from that of the vehicle control. Bruggeman et al. (2005) evaluated reproductive performance of domestic chicken hens that were exposed to TCDD *in ovo*, and although physiology was affected, reproductive performance was not. Because of these findings, this toxicity profile focuses on studies in which egg mortality is the endpoint.

2.1.4.2 Reproductive Effects and Toxicity to Embryos

Many studies have been conducted to address the toxicity of TCDD and dioxin-like PCBs to bird eggs and developing embryos, but not all of the available studies can be used for risk assessment (Section 1.4). Given the complexity of the literature describing avian toxicity in terms of the variety of species, the ranges of results, and the methods used for selection of studies, USEPA (2003) was used as a starting point for the literature evaluation needed to identify TRVs expressed as egg concentrations for this BERA. The literature presenting information for use in developing ingestion TRVs is much smaller. Both are discussed below, as are the selected TRVs.

2.1.4.2.1 Egg Tissue TRVs

Use of egg-exposure based TRVs is the recommended risk assessment approach by both TN & Associates (2002) and USEPA (2003). USEPA (2003) provides a compilation of results of toxicity tests in which exposures as concentrations in eggs were documented, building on the detailed literature review conducted by TN & Associates (2002) for USEPA's Office of Research and Development. They used their aggregation of data to prepare SSDs for birds. Both laboratory and field studies were compiled by USEPA (2003). A paper was only selected for use in USEPA's (2003) analysis if it included all of the following:

- Evaluation of more than one quantitative dose or exposure level. Studies evaluating only one dose or exposure level were considered to have too much uncertainty.
- One or more quantifiable toxicological endpoint.
- Appropriate statistical tests showing significant changes in response with changes in dose or exposure levels.
- Evaluation of the potential for co-contaminants to affect results (for field studies).

USEPA's (2003) compilation of TRVs expressed as TCDD (or TEQ) concentrations in eggs includes NOAELs for developmental impairment from laboratory studies ranging from 66 ng TEQ/kg egg for the chicken to 50,000 ng TEQ/kg egg for several other bird species, including two gull species, the Graylag goose, and the goldeneye (a duck). Corresponding LOAELs range from 150 to 4,400 ng TEQ/kg egg. Not all of these studies were used for developing egg tissue TRVs, as discussed below.

Finally, TCDD or other toxicants can be injected into bird eggs in one of several ways: into the air cell in the egg, into the albumin (white) of the egg, or into the yolk. Nosek et al. (1992a) injected radiolabeled TCDD into laying pheasant hens once a week for 10 weeks; hens were induced to lay eggs during the final two weeks of exposure. The first 15 eggs from each hen were collected, and the yolk and albumin were separated. Nosek et al. (1992a) report that greater than 99 percent of the TCDD-derived radioactivity translocated from hens to eggs was found in the yolk; none was detected in albumin. Some authors have noted that injection into the air cell may contribute to egg suffocation, when the oily material used in dosing impedes the transfer of oxygen to the embryo (Henshel et al. 1997a). For these

reasons, TRVs reported by laboratory studies in which dosing occurred by injection into the yolk are preferred because they are considered to be more biologically realistic.

To summarize, controlled laboratory studies meeting USEPA (2003) criteria and using injection into yolks or maternal transfer as the means of administration and in which egg mortality was the endpoint were preferred for this risk assessment. Although there are a number of studies of the toxicity of PCB126 and other PCB congeners to birds, because of the inter- and intraspecies variability, and uncertainties about Van den Berg et al.'s (1998) TEFs for birds (e.g., Cohen-Barnhouse et al. 2011), Integral also used only studies in which TCDD was the toxicant of interest.

USEPA (2003) cites data from the following studies of TCDD toxicity in its compilation of egg tissue TRVs, which was the starting point for this evaluation:

- Henshel et al. 1997a. The Relative Sensitivity of Chicken Embryos to Yolk- or Air-Cell-Injected 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin.
- Powell et al. 1996. Effects of 3,3',4,4',5-Pentachlorobiphenyl (PCB 126) and 2,3,7,8-Tetrachlorodibenzo-*p-D*ioxin (TCDD) Injected into the Yolks of Chicken (*Gallus domesticus*) Eggs Prior to Incubation.
- Powell et al. 1997a. Effects of 3,3',4,4',5-Pentachlorobiphenyl (PCB 126), 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD), or an Extract Derived from Field-Collected Cormorant Eggs Injected into Double-Crested Cormorant (*Phalacrocorax auritus*) Eggs.
- Powell et al. 1997b. Organochlorine Contaminants in Double-Crested Cormorants from Green Bay, Wisconsin: II. Effects of an Extract Derived from Cormorant Eggs on the Chicken Embryo.
- Powell et al. 1998. Effects of 3,3',4,4',5-Pentachlorobiphenyl and 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin Injected into the Yolks of Double-Crested Cormorant (*Phalacrocorax auritus)* Eggs Prior to Incubation.
- Nosek et al. 1992b.[3] Toxicity and Reproductive Effects of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin in Ring-Necked Pheasant Hens.

[3] USEPA (2003) cites Nosek et al. (1992b), but the data appear to be from Nosek et al. (1993). Integral used results from both studies.

- Henshel et al. 1997b. Brain Asymmetry as a Potential Biomarker for Developmental TCDD Intoxication: A Dose-Response Study.
- Walker et al. 1997. Expression of the Aryl Hydrocarbon Receptor (AhR) and AhR Nuclear Translocator during Chick Cardiogenesis Is Consistent with 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin-Induced Heart Defects.

Henshel et al. (1997b), Powell et al. (1997b), and Walker et al (1997) were not used because the study endpoints were not relevant to evaluation of ecological risks. Integral also reviewed and used data from two additional studies exposing eggs via yolk injection or maternal transfer by Nosek et al. (1992a; 1993). Results of three studies in which the TCDD was administered via injection into the air sac or albumin are also summarized to provide valuable perspective on the yolk injection studies: Cohen-Barnhouse et al. (2011), Henshel et al. (1997a), and Nosek et al. (1993).

Finally, the majority of literature on dioxin toxicity to birds reports information on field collected eggs. USEPA (2003) also compiled results from field studies and analyzed them separate from laboratory studies. Results of field studies are uncertain because of potential for effects of co-contaminants including pesticides, PCBs, and chemicals not measured by investigators, and therefore most field studies are not appropriate for use in risk assessments to define effects levels. However, no-effects levels derived from field studies can provide some perspective on Site-specific exposure estimates. A subset of the available field studies and the NOAECs for eggs that they report is also summarized.

Laboratory Based Yolk-Injection Studies

Studies by Nosek et al. (1992a,b; 1993) and Powell et al. (1997a, 1998) on common pheasants and double crested cormorants in which yolks were injected with TCDD form the basis for the TRV for bird eggs used in this risk assessment. Their results are summarized in Table B-6, along with two chicken studies included for perspective. These studies can be summarized as follows:

- Nosek et al. (1992b) conducted experiments with ring-necked pheasant hens, dosing individual hens weekly with intraperitoneal injections of 0, 0.01, 0.1, and 1.0 µg TCDD/kg bw per week. Significant reductions in survivorship, reduced egg

production and an increase in cumulative egg mortality were observed in hens receiving 1.0 µg/kg-week, or a cumulative dose of 10 µg/kg bw, but not at lower doses. The effect observed at this dose was 100 percent egg mortality. Nosek et al. (1992b) estimate that the cumulative dose to individual hens resulting in 50 percent egg mortality was 4.5 µg/kg bw. This value and data from two other studies were used to derive the egg TRVs from the Nosek et al. (1992b) study: In the first study, Nosek et al. (1992a) injected 0.1 µg/kg bw radiolabeled TCDD into pheasant hens once a week for 10 weeks and found that about 1 percent of the total dose to hens was translocated to each of the first 30 eggs laid. Thus, the approximate dose to eggs resulting in 50 percent egg mortality (from Nosek et al. 1992b) is 45 ng/egg. In the second study, Nosek et al. (1993) report a mean egg weight for their pheasants of 30.5 g. This information was used to calculate a LOAEL for eggs from the Nosek et al. (1992b) of 1,477 ng/kg ww egg. The NOAEL from Nosek et al. (1992b) was similarly derived using a cumulative dose to hens of 1 µg/kg bw, resulting in 10 ng/egg or 328 ng/kg ww (Table B-6).

- Nosek et al. (1993) injected TCDD into yolks of pheasant eggs (and into albumin, below) at doses of 0, 10, 100, 1,000, and 10,000 ng /kg egg on Day 0 of embryonic development. An increase in mortality over the control group of 20 percent of eggs was observed at 1,000 ng/egg, and 98 percent mortality was observed at the highest dose. For the purposes of risk assessment, 1,000 ng/kg egg is considered the LOAEL from this study. The LD_{50} (the egg concentration at which 50 percent of organisms die) was calculated as 2,150 ng/kg egg.
- Powell et al. (1997a; 1998) conducted studies using eggs of double-crested cormorant collected from a remote area of Canada. The authors acknowledge the presence of TCDD and PCBs in cormorant eggs from that region, but performed a test to see whether these baseline residues would influence their studies. They injected an extract of TCDD and PCBs from untreated eggs into test eggs (Powell et al. 1997a) and found no effect on egg mortality, so the baseline level of contaminations is not likely to interfere with the experiments involving higher doses. In the 1997 study, these authors injected yolks of cormorant eggs with 60, 250, 1,000 and 4,000 ng TCDD/kg egg. Egg mortality was significantly elevated over controls only at the highest dose. In the 1998 study, egg yolks were injected with 1,300, 5,400, 10,700 and 11,700 ng/kg egg. At concentrations of 5,400 ng/kg and higher, mortality was significantly elevated

over controls. However, even at the highest dose, about 15 percent of eggs survived, demonstrating the low sensitivity of cormorants to TCDD relative to other species. This series of studies has been criticized for high control mortality, but such is to be expected with wild-captured eggs. The authors demonstrated that low levels of TCDD and PCBs in the collected eggs had no effect on the outcome, and even if it did, the result would be a conservative TRV. Otherwise, these studies are robust, demonstrating clear dose-response relationships, and are considered valuable because the eggs were from a wild stock.

Data for these four laboratory studies form the basis for derivation of the TRV for bird eggs used in this risk assessment (Tables B-6 and B-7). There were insufficient data for derivation of an SSD for bird eggs. The geometric means of NOAELs and LOAELs from the two studies for each of the two species were calculated, and the geometric mean of the resulting two geometric means were calculated to derive the TRVs for this risk assessment. The resulting NOAEL and LOAEL for TEQ_B in bird eggs, rounded to two significant figures are 450 and 2,400 ng/kg egg, respectively. The within-species geometric means were calculated first to minimize the influence of any one species on the final TRVs.

Because domestic chickens are clearly more sensitive than all other species, results from studies with chickens were not included in derivation of the TRVs. However, there are two yolk injection studies with domestic chickens summarized in Table B-6:

- Henshel et al (1997a) injected yolks of chicken eggs with 10, 30, 60, 100, 300, and 1,000 ng/kg egg ww and observed significantly elevated mortality (100 percent) at 300 ng/kg. The sample size in this study was fairly small per treatment, but a dose-response relationship was observed.
- Powell et al. (1996) also injected eggs of domestic chickens at 0, 40, 80, 160, 320, and 640 ng/kg egg. A statistically significant increase in egg mortality was observed at 160 ng/kg.

Results of these two studies are included in Table B-6, and the geometric mean NOAEL and LOAEL were calculated to be 89 ng/kg and 219 ng/kg, respectively. Using all three geometric means (chickens, pheasants, and cormorants) yields an overall geometric mean NOAEL rounded to two significant figures of 260 ng/kg and LOAEL of 1,100 ng/kg for bird

eggs. These are considered in the uncertainty assessment to evaluate exposure estimates for birds.

Laboratory-Based Albumin or Air Cell Injection Studies

Studies in which eggs are exposed via injection to the albumin or the air sac are summarized in Table B-8. Results generally agree with those of the yolk injection studies, two of which are discussed above (Henshel et al. 1997a; Nosek et al. 1993), although chickens seem somewhat less sensitive using these data. In the third, Cohen-Barnhouse et al. (2011) was a detailed study evaluating relative sensitivity among bird species to developmental effects. The three bird species they studied are considered to have widely different sensitivities on the basis of enzyme induction. While this study presents a lot of important information, including an indication that 2,3,4,7,8-PeCDD and TCDF are more toxic to pheasants and quail eggs than are other congeners, this summary is focused on egg mortality. Results indicate that quail are considerably less sensitive than the other bird species discussed in this toxicity profile, with a NOAEL for egg mortality of 3,542 ng/kg, and an LOAEL of 9,015 ng/kg. These data were not included in calculation of geometric means because they are not yolk-injection data, but suggest that the TRVs derived for this risk assessment are reasonably conservative, in light of uncertainties.

Field Studies

In field studies reviewed by USEPA (2003), NOAELs for developmental effects range from 5 ng TEQ/kg egg for the wood duck to 1,440 ng TEQ/kg egg for the Caspian tern. Among all of the studies available on this topic, this summary is a selection of recently published literature including one from the area of the Site, for species selected as receptors for this risk assessment, and studies in which the authors performed a risk assessment for birds. The following studies are summarized in Table B-9:

- Frank et al. (2001) evaluated concentrations of multiple persistent organic compounds in waterbird eggs in the Galveston Bay area. In addition to several areas sampled within the Galveston Bay area, two reference areas were included for comparison of chemical levels in eggs and adverse health effects. Eggs were collected from three bird species: neotropic cormorants (n=28 eggs from four sites; n=18 eggs from two reference sites), black-crowned night herons (n=9 eggs from one site), and great egrets

(n=7 eggs from one site). The collected eggs were evaluated for concentrations of pesticides, PCBs, dioxins, and furans. Egg extracts were also evaluated for AhR activity using a bioassay and examined for developmental abnormalities. TEQs from eggs collected within the Galveston area site ranged from 166 to 452 ng/kg compared to a TEQ of 67 ng/kg for the cormorant egg from one of the reference areas (Telfair Island). TCDD contributed 26 to 51 percent of calculated TEQs with the majority of the remainder being attributable to PCB 126. No deformities or abnormalities in embryos were detected at all sites investigated, suggesting a NOAEL of 452 ng/kg for neotropic cormorant, 376 ng/kg for the black crowned night heron, and 240 ng/kg for the great egret.

- Woodford et al. (1998) examined the survival, growth, and behavior of breeding ospreys exposed to TCDD in Wisconsin. The third eggs of freshly laid clutches were collected over different years (total n=18) from two contaminated sites. Eggs were also collected at two reference sites over three different years (total n=15). Eggs collected from one of the sites had TCDD levels of 29 to 162 ng/kg wet weight while the reference areas had a reduced range from below detection limits to 23.8 ng/kg. Despite the difference in TCDD levels, egg exchange experiments between the affected sites and reference areas showed no significant differences in egg hatching or fledgling rates. A difference (p=0.03) was noted in growth of chicks with the group from the contaminated sites affected. Using the reproductive endpoints of egg hatching and chick fledgling rates suggests a NOAEL of ≥136 ng/kg.
- Custer et al. (2010) investigated the nest and egg success of spotted sandpipers by weekly surveillance of nests on the Hudson River (24 nests) and less frequently in two reference sites (18 nests). Eggs were also collected for chemical analysis (Hudson River, n=13; reference areas, n=5) of PCBs, dioxins, furans and organochlorine pesticides. $TEQ_{DFP,B}$ ranged from 75 to 6,540 ng/kg wet weight on the river and from 8 to 56 ng/kg wet weight for the two reference areas. TEQs at all locations were dominated by PCBs with dioxin and furans contributing fewer than 10 percent of the $TEQ_{DFP,B}$. Results were analyzed using the small sample variant of Akaike Information Criterion to test if nest and egg success was related to TEQ concentration. Models that predicted nest survival and egg success as functions of contaminant levels were poorly distinguished from models that presume no such associations indicating that the contaminant concentrations did not have a sufficient identifiable statistical

relationship with reproductive success. Using the geometric mean $TEQ_{DFP,B}$ concentrations, a NOAEL of 732 ng/kg wet weight is inferred for sandpipers for the Hudson River site.

- Elliott et al. (2001) summarize concentrations of dioxins and furans in great blue heron eggs collected before during and after major changes to the pulp and paper sector in British Columbia, Canada. During the 16-year study, eggs were collected from 21 rookeries either during the egg-laying or incubation period depending on the year. A linear relationship was established between prey fish species and heron egg contamination levels indicating that local dietary uptake was a key exposure route. $TEQ_{DF,B}$ concentrations were elevated through the 1980s at levels sufficient to produce embryotoxicity (reduced chick size, increased brain asymmetry, elevated EROD activities) but decreased markedly in the early 1990s due to changes in bleaching practices of local pulp mills. Although heron $TEQ_{DFP,B}$ concentrations declined after 1990, levels remain near 200 ng/kg ww at one site due to the persistent presence of PCBs. No gross abnormalities or deleterious effects on the number of fledglings were observed at the reference site, Nicomekl River, from which an NOAEC for great blue heron of 207 ng $TEQ_{DFP,B}$/kg ww is estimated.

The field studies described above provides a range of NOAEC values from 136 to 732 ng/kg ww, which cover multiple bird species including the receptors surrogates for the Site. The highest LOAEL value reported in the field (1,700 ng/kg ww) for sandpipers is associated with reduced hatching although there was no effect on nestling survival and growth. In a risk assessment study using multiple lines of evidence (Fredricks et al. 2011), TRVs of 710 and 1,000 ng/kg ww were used as NOAECs for the house wren and eastern bluebird, respectively. The use of these TRVs was supported by field observations of no significant population level effects on reproduction at concentrations below these levels. The geometric mean of NOAECs from three studies on bird receptors present at the site is 411 ng/kg ww (Custer et al. 2010; Elliott et al. 2001; Frank et al. 2001). These field values are consistent with the egg tissue TRVs derived from laboratory yolk injection studies (Table B-6).

2.1.4.2.2 Ingestion Rate TRVs

Research supporting development of NOAELs and LOAELs for birds expressed as an ingested dose of TCDD is rare. For dietary TRVs, only one laboratory study was found in which feeding was the route of administration, conducted on the domestic chicken (Schwetz et al. 1973). USEPA's review of literature from 1981 to 1997 (TN & Associates 2002) also did not identify any studies of acceptable quality reporting ingested doses, and concludes that egg tissue data provide the best means for assessing exposures to birds and evaluating risk (USEPA 2003).

Schwetz et al. (1973) fed 3-day old white leghorn chicks 2,3,7,8-TCDD mixed into food for 21 days using a standardized assay for assessment of chick edema. This study reports a LOAEL for several effects, including chick edema and reduced chick survival of 1 µg/kg diet, with a NOAEL from the same study of 0.1 µg/kg diet. This study was not selected for use in this BERA because some methodological details and test conditions were not included, and the authors report only nominal concentrations in feed, resulting in uncertainty about the actual doses.

Nosek et al. (1992b) conducted experiments with ring-necked pheasant hens, dosing individual hens weekly with intraperitoneal injections of 0, 0.01, 0.1, and 1.0 µg TCDD/kg bw per week. Significant reductions in survivorship reduced egg production and an increase in cumulative egg mortality were observed in hens receiving 1.0 µg/kg per week, or a cumulative dose of 10 µg/kg bw. Affected hens exhibited wasting syndrome. TRVs expressed as a daily dose can be derived from this study. A LOAEL for effects on fertility and hatching success of 1.0 µg/kg bw per week was converted to LOAEL expressed as a daily ingestion rate of 140 ng/kg-day (Nosek et al. 1992b). The dosing regimen was based on orders of magnitude differences and adverse effects were not observed at the next lowest dose, the NOAEL (14 ng/kg-day). Although test subjects were not fed the test chemical, this approach to deriving an oral TRV is appropriate because the exact doses to birds are known, the study is thoroughly reported and robust, and the result is conservative because it assumes 100 percent uptake from an ingested dose, likely an overestimate (Nosek et al. 1992a). It is also in general agreement with Schwetz et al. (1973) despite using different species and routes of administration. Several other reviewers also use results of Nosek et al. (1992b) to derive oral TRVs for use in risk assessment (Sample et al. 1996; Fredricks et al. 2011; Windward 2011a).

2.1.5 Mammals

Toxicity of dioxins and furans in mammals is mediated through AhR, as it is in fish and birds. Exposure of mammals to dioxins and furans is associated with adverse effects on reproduction and development, and the sensitivity of mammals to TCDD toxicity is highly variable. 2,3,7,8-TCDD toxicity in mammals may be characterized by loss of body weight and death. Atrophy of the thymus is consistently a manifestation of 2,3,7,8-TCDD toxicity in mammals, and suppression of thymus-dependent cellular immunity, particularly in young animals, may contribute to their death. Early life stages, including the fetus and newly born pup/kit, appear to be the most sensitive to dioxin toxicity, and maternal exposure can result in increased frequencies of stillbirths. Acute toxicity studies with 2,3,7,8-TCDD have shown marked differences among species; up to a factor of 8,400 between the single oral LD_{50} dose for the guinea pig (the most sensitive mammal) and the hamster (Eisler 1986a).

Unlike fish and birds, only one form of AhR has been shown to exist in mammals. Differences in sensitivity to dioxins and furans may therefore be a function of species-specific toxicokinetic and toxicodynamic factors. The majority of mammalian studies addressing toxicity of dioxins and furans have used common laboratory species (e.g., rat and monkey). Mink have also been the subject of several dioxin and furan exposure studies, and recent literature clearly indicates that 2,3,7,8-TCDF is much less toxic to mink than would be predicted by the mammalian TEF for this congener (Zwiernik et al. 2009). The limited range of mammalian test species argues for caution in extrapolating many of the results to other mammalian wildlife species, which may differ substantially in their life history and ecology from tested animals.

Given the large literature for toxicity of dioxins and furans on rodents and other taxa commonly used in evaluating potential for effects on people, the literature review supporting development of TRVs for mammals is not comprehensive. Studies that are summarized below address survival, growth, or reproduction in mammals.

2.1.5.1 Effects on Growth

In a three-generation reproductive toxicity study of rats, body weight and thymus weight were significantly reduced in third generation female pups in the 0.01 µg TCDD/kg bw-day dietary exposure group (Murray et al. 1979). There were no significant changes in body mass of captive mink exposed to increasing concentrations of dietary TCDF, PeCDF, or a mixture of the two, of up to 9.5 ng TEQ/kg bw-day for up to 6 months (Moore et al. 2009).

Some significant but transient decreases in mink kit mass have been observed, particularly in female kits, at predicted maternal liver concentrations of 36 and 980 ng TCDF/kg liver ww (3.6 and 9.9 ng TEQ/kg liver ww) (Zwiernik et al. 2009). However, in a 3-year field study of trapped mink, adverse effects of a mixture of dioxins and furans on sex ratio, body weight, length, liver weight, and baculum length were not seen at estimated dietary concentrations of 31 ng TEQ/kg ww, a value that was primarily driven by furan content; tissue congener analysis was not reported (Zwiernik et al. 2009). Given Zwiernik et al.'s (2009) finding that TCDF may not be as potent in mink as predicted by the World Health Organization's 2005 TEF for that congener, these studies do not provide a reliable indicator of effects thresholds expressed as TEQ because TCDF is a large component of the exposure in these studies.

2.1.5.2 Reproductive Effects

Several studies document the effects of 2,3,7,8-TCDD on reproduction in mammals. The most applicable to this BERA is a three-generation study with Sprague-Dawley rats exposed continuously to diets with 0.001, 0.01, and 0.1 µg of 2,3,7,8-TCDD/kg bw-day (Murray et al. 1979). No effects were seen at any dose in the first generation rats administered contaminated feed for 90 days, but 0.1 µg /kg bw-day resulted in reduced survival of their neonates. Several reproductive effects were observed in the second generation rats receiving 0.01 µg/kg bw-day, including reductions in rat fertility, decrease in litter size, and significant reductions in the number of pups born alive (Murray et al. 1979). In the same study, pup survival was reduced at the dietary exposure level of 0.01 µg/kg bw-day in second and third-generation rats, but not in the first. Evidence of reproductive toxicity in monkeys is also available: the number of viable offspring born was reduced in groups of female monkeys exposed to 25 ppt TCDD in the diet prior to mating and during gestation and lactation relative to control (Bowman et al. 1989; Schantz et al. 1992).

For mink, older studies generally only report the dose which was lethal to 50 percent of specimens (LD_{50}), or they are feeding studies with fish contaminated with chemicals other than the target dioxin-like compounds, but the data suggest mink are sensitive to toxicity of some dioxin-like compounds. Hochstein et al. (1998) evaluated a range of endpoints in mink fed TCDD at doses ranging from 0.000055 to 3.0 µg /kg-day, resulting in a LOAEL for mortality following 125 day exposure of 0.054 µg /kg-day. A more recent study found no effect on fertility, percentage of kits born alive, or kit survival relative to control in mink administered TCDF at concentrations of 26 or 240 ng TEQ/kg ww diet for three weeks prior to breeding through birth and weaning (Zwiernik et al. 2009). These results illustrate that TCDF is not as toxic in mink as would be expected from the mammalian TEF for that chemical.

Exposure to dioxins and furans during gestation and/or lactation can result in effects in the developing fetus and offspring. However, there is comparatively little empirical evidence for toxicity to the developing female; studies with rats almost exclusively evaluate effects in male pups. Male offspring of exposed mothers exhibit reductions in reproductive organ weights, reduced steroidogenesis, and reduced sperm count (Ohsaka et al. 2002; Hamm et al. 2003; Ikeda et al. 2005; Mutoh 2006). Adverse effects of TCDD on sperm production have been observed in rats (Hamm et al. 2003; Faqi et al. 1998), and effects on sperm production may occur at a lower exposure level (0.25 ng TCDD/g testicular tissue) than the exposure level associated with reduced fertility. Although this is a reproductive effect, reduction in sperm production may have no material effect on wild mammal populations.

Other studies with mammals are available for interpreting exposures to dioxin-like compounds at the Site include Khera and Ruddick (1973) and Kociba et al. (1978). Khera and Ruddick (1973) measured litter size and pup weight in rats exposed to 2,3,7,8-TCDD for 10 days during the period of gestation, reporting a LOAEL of 0.25 µg TEQ/kg-day and a NOAEL of 0.125µg TEQ/kg-day. Kociba et al. (1978) exposed rats to 2,3,7,8-TCDD for 2 years, observing increased mortality in females dosed with 0.1 µg TEQ/kg-day, and no effect on female mortality at 0.01 µg TEQ/kg-day. Because the first of these was a short-term study, and the second did not measure reproductive effects, neither one was used to derive TRVs for this BERA.

In light of uncertainties about the TEFs for mink, because mink are not a receptor at the Site, and because results of the multiple-generation study with rats by Murray et al. (1979) was the only chronic reproductive study, results from Murray et al. (1979) were used to interpret estimated exposures of mammalian receptors for this BERA. The NOAEL of 0.001 µg /kg-day and the associated LOAEL of 0.01 µg/TEQ/kg-day from Murray et al. (1979) were selected as TRVs for mammals (Table B-10).

2.2 Polychlorinated Biphenyls

The toxicity of PCBs to fish, mammals, and birds is relatively well studied. TRVs for exposure of these receptors to PCBs are available as effects levels for three forms: 1) Aroclors (USEPA 2004); 2) total PCBs; and 3) 2,3,7,8-TCDD TEQs. Each of these TRV forms has advantages and disadvantages. Although use of the TEQ approach allows risk analysts to evaluate cumulative exposures and toxicity to multiple compounds, any effects not mediated by the AhR pathway are not accounted for by this method.

Field and laboratory data suggest that many of the toxic effects caused by planar PCBs are mediated subcellularly by AhR, the same receptor responsible for mediating dioxin toxicity. This receptor is involved in the translocation of PCBs into the nucleus and their subsequent binding to AhR (Safe 1991). Because of similar mechanisms of action through binding to AhR, the signs of PCB126 toxicity, for example in lake trout early life stages, are similar to those shown by TCDD, and include yolk-sac edema, multifocal hemorrhages, craniofacial malformation, and mortality (Zabel et al. 1995).

However, although recent work has suggested that while the TCDD-like congeners act by a common mechanism (i.e., AhR), the combined effects of TCDD with the coplanar PCB congeners may not be additive (Walker et al. 1996) because competition for binding sites among the dioxin-like compounds may result in the less potent congeners being the more important driver of response. Despite this uncertainty, the additive model continues to be acceptable for assessing risk because deviation from additivity has been estimated to be within an accepted tenfold range (Walker et al. 1996). TRVs used for interpretation of

exposures to dioxin-like PCBs are those discussed and presented in the section on dioxins and furans.

PCBs can produce a variety of responses in organisms and act as neurotoxicants, hepatotoxicants, immunotoxicants, and carcinogens (Safe 1991; Shain et al. 1991; Giesy and Kannan 1998). While sensitivity and responses tend to be species-specific, general responses include lethality, reproductive and/or developmental toxicity, neurotoxicity, hepatic lesions, tumor promotion, suppression of the immune system, and induction of drug-metabolizing enzymes (McFarland and Clarke 1989; Safe and Phil 1990; Eisler and Belisle 1996; Giesy and Kannan 1998). In vertebrates, PCBs induce the cytochrome P450 metabolic enzyme system (Eisler and Belisle 1996). The degree of metabolic breakdown is dependent on the degree of chlorination and the spatial arrangement of chlorine atoms. As the number of chlorine atoms in the PCB molecule increases and the number of unsubstituted adjacent carbon atoms decreases, metabolic transformation decreases. PCB elimination is limited and PCBs bioaccumulate in organisms and biomagnify within food chains.

Of the 209 possible PCB congeners, research has indicated that as much as 75 percent of tissue burdens of PCBs in invertebrates, fish, birds, and mammals consist of only 25 congeners (McFarland and Clarke 1989). These congeners with the greatest likelihood for bioaccumulation and toxicity are the planar non-, ortho-, or mono-ortho-substituted PCBs, which chemically resemble and toxicologically behave similarly to the 2,3,7,8-substituted PCDFs and PCDDs (Walker and Peterson 1991). Specifically, several lines of testing have implicated the planar PCB congeners 77, 81, 126, and 169 as major contributors to the toxicity of PCB mixtures (Ankley et al. 1991).

This section describes TRVs expressed as "total PCBs" based on toxicity studies using Aroclors. Although total PCBs may be calculated as the sum of congeners, the sum of a subset of congeners, or the sum or Aroclors, selection of TRVs expressed as an Aroclors may sometimes be necessary because TRVs expressed as a sum of congeners are not available. In these cases, the exposures based on sum of Aroclors or sum of congeners is compared to a TRV expressed as an Aroclor, a conservative approach.

TRVs expressed as total PCBs are included for use in the BERA. Risks evaluated using total PCBs as the exposure metric are considered separately (not additively or cumulatively) from risks associated with the dioxin-like toxicity of PCBs. Site-specific data allow for estimates of exposure to total PCBs by most receptors, and include: all 209 congeners in tissue samples collected for the Site; Aroclors and dioxin-like congeners at 12 soil stations, and Aroclors in surface and subsurface soils for the southern impoundment area; surface sediment data for Aroclors and dioxin-like congeners collected for the remedial investigation and a few samples at one location (under the I-10 bridge) collected by TCEQ; and surface water data for all 209 congeners in a few samples collected by TCEQ in one location under the I-10 bridge.

There is uncertainty associated with the use of Aroclor 1254 toxicity information in combination with total PCBs as the exposure metric. The mixture of PCB congeners in sediments and tissue at the Site may not reflect the same congener composition as Aroclor 1254. Nevertheless, the assessment approach should be protective because Aroclor 1254 is expected to be among the Aroclors most toxic to birds and mammals based on extrapolation of comparative studies of Aroclors in aquatic organisms (Nebeker and Puglisi 1974; Mayer et al. 1977; Johnson and Finley 1980). Moreover, dechlorination of PCBs by natural processes at the Site would likely lead to mixtures with toxicity less than or equal to Aroclor 1254, because Aroclor 1254 is a mixture of highly chlorinated PCBs, which generally have relatively high toxicity. PCB toxicity to birds and mammals is addressed in the BERA with approaches based on both total PCB exposures and $TEQ_{P,B}$ and $TEQ_{P,M}$ exposures, respectively. The finding of negligible risk for the Site based on $TEQ_{P,F}$, $TEQ_{P,B}$, and $TEQ_{P,M}$ supports the overall conclusion of negligible risk to fish, birds, and mammals from PCBs, improving confidence in similar conclusions for analyses based on total PCBs.

2.2.1 Fish

The effects of PCBs on Great Lakes fish and wildlife have been extensively studied. PCB-induced reproductive impairment has been demonstrated for several fish species (Ankley et al. 1991; Mac 1988; Walker et al. 1991a; Walker et al. 1991b; Walker and Peterson 1991; Williams and Giesy 1992). Generally, the most sensitive endpoints for effects of PCBs in fish are early life-stage survival and recruitment where exposure has resulted

from transfer of PCBs from maternal tissue to eggs (Eisler and Belisle 1996; Walker et al. 1996). Whole-body concentrations of PCBs in adult fish that are commonly found in the environment do not generally result in death (Eisler and Belisle 1996). This is consistent with numerous field studies evaluating PCB fish tissue concentrations and adverse effects summarized by Niimi (1996). Based on several field studies, lethal body burden concentrations have been estimated at greater than 100 mg/kg for young fish and greater than 250 mg/kg for older fish (Niimi 1996).

Numerous studies report TRVs as residues in tissue of fish administered PCBs through water only, food only, or water and food combined. The tissue-based NOAEL and LOAEL for this risk assessment were developed primarily from the literature. The derivation of TRVs focused on the measurement endpoints related to survival, growth, and reproduction and, at USEPA's request, data for freshwater fish species were not included. The methodology used to combine data when deriving the tissue-based TRVs was analogous to that used to derive USEPA EcoSSLs for soils.

2.2.1.1 Derivation of NOAEL for Total PCBs in Whole Fish

There were several studies reported in the literature or used by state or federal agency ecological risk assessments that reported NOAEL values for fish expressed as whole-body concentrations that were included in calculation of the NOAEL for this BERA.

- Hansen et al. (1973) exposed female sheepshead minnow (*Cyprinodon variegatus*) to Aroclor 1254 in water using a flow-through system. Fish were exposed for 28 days to control water or five nominal concentrations of Aroclor1254 in water (0.1, 0.32, 1.0, 3.2, and 10 µg/L). All fish survived and egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality. Survival of fry to 1 week of age was 77 percent for eggs from adults from the 0.32 µg/L concentration in water treatment (average 9.3 mg/kg in tissue of females), as compared to 95 percent survival of fry from control adults and 97 percent survival of fry from adults from the NOAEL treatment (0.1 µg/L; average 1.9 mg/kg in tissue of females). This study was used to derive the tissue-based NOAEL and LOAEL of

1.9 mg/kg and 9.3 mg/kg, respectively, for both the Hudson River Revised BERA (USEPA 2000b) and Onondaga Lake BERA (NYSDEC 2002).[4]

- Bengtsson (1980) exposed adults of the common minnow (*Phoxinus phoxinus*; also called the Eurasian minnow) to the PCB mixture Clophen A50. Clophen A50 contains 50 percent chlorine by weight and has similar physicochemical parameters to Aroclor 1248. Fish were exposed to a control diet, or a diet fortified at three PCB levels (20, 200, and 2,000 mg/kg) for 40 days, and then monitored for a total of 300 days. Fish were subsampled at several times during this period and their whole-body PCB concentrations were quantified. Growth, reproduction, and behavioral effects (i.e., swimming) were monitored during this period. There was no apparent impact on hatchability of the ova from exposed adults for average whole-body tissue concentrations up to 15 mg/kg ww (this corresponded to the 200 mg/kg diet exposed fish). The value of 15 mg/kg ww represents the NOAEL.
- Westin et al. (1983) fed striped bass (*Morone saxatilis*) larvae PCB-contaminated brine shrimp (*Artemia* spp.) from yolk sac absorption to either 10 or 20 days. Larvae were left to deplete the yolk sac for the first 10 days, and then fed the PCB-contaminated shrimp for 20 days. Larvae were subsampled at test initiation, at post-yolk sac (day 10), after 10 days of feeding (day 20), and after 20 days of feeding (day 30), and analyzed for PCBs. Survival and growth were monitored during the study period. There was no apparent impact on survival or growth in the treatment, and PCB concentrations in tissue were found to decline over the monitoring interval, which was attributed to growth dilution. The highest post-yolk sac whole-body tissue concentration was in the larvae fed contaminated shrimp for 10 days, and was 4.4 mg/kg ww. This concentration is included as a NOAEL, as required by USEPA in comments on the draft of this report (Appendix F). Uncertainty associated with this result is due to the lack of any observed effect (i.e., the NOAEL is unbounded) and the fact that the study period encompassed a period of rapid larval growth and consequent dilution of the dose into the larval tissues. These uncertainties result in a NOAEL that is lower than it would be if the test organisms were not undergoing a high rate of growth at the time of dosing, and that is not clearly representative of any particular life stage.

[4] This paper was cited as "Hansen et al. (1974)" in USEPA (2000) and NYSDEC (2002), but the correct publication year was 1973.

2.2.1.2 Derivation of LOAEL for Total PCBs in Whole Fish

Two of the studies that are discussed above (Bengtsson 1980 and Hansen et al. 1973) also reported LOAEL values. In the Bengtsson (1980) study, the individuals with the highest exposure (2,000 mg/kg in their diet) had an average whole-body total PCB concentrations of 170 mg/kg. Eggs of these fish exhibited decreased hatchability, so 170 mg/kg is the LOAEL derived from Bengtsson (1980). The Hansen et al. (1973) study was used in the draft Hudson River BERA (USEPA 1999b) to derive the TRV, but was not used for the Hudson River Revised BERA (USEPA 2000b). The Hansen study LOAEL was 9.3 mg/kg.

An additional study by Orn et al. (1998) was included in calculation of the LOAEL, as required by USEPA in comments on the draft (Appendix F). Orn et al. (1998) purchased adult zebrafish (*Danio rerio*), and after 4 weeks of acclimatization, exposed the fish to a mixture of 20 selected PCB congeners at three dose levels in feed for 13 weeks. The reproduction study was initiated following 9 weeks of exposure. Survival, growth, and reproduction were monitored during the study, along with various histopathological endpoints (e.g., the liver somatic index) that required removal of the subject organs from the dosed females. A reduction in the number of eggs per female and reduced larval survival were observed in the high dose group, resulting in a LOAEL of 2.7 mg/kg, the concentration in the females after livers and ovaries were removed. Potential confounding factors in this study include uncertainties about the representativeness of the 20 selected PCB congeners of mixtures to which fish may be exposed, and the fact that ovaries and liver were removed prior to tissue PCB analysis. As a result of the organ removals, it is likely that the LOAEL is a substantial underestimate because PCBs accumulate in ovary and liver tissue of fish.

The recommended NOAEL and LOAEL values are 5.0 and 16 mg/kg ww in whole body fish tissue, respectively (Table B-11). These TRVs are the geometric mean TRVs derived from the studies accepted for TRV development as described above and summarized in Table B-12.

2.2.2 Reptiles

The possibility that exposure of reptiles to PCBs and related elevated concentrations in turtle eggs could result in deformities in developing turtles is discussed in Section 1.3 and 2.1.3.

Exposure-response relationships to describe effects of PCB exposure on turtles and other reptiles have not been developed, and acceptable TRVs for interpreting exposures to reptiles on the Site are not available.

2.2.3 Birds

Effects of ingested PCBs on birds include disruption of normal patterns of growth, reproduction, metabolism, and behavior. PCB-induced reproductive impairment has been demonstrated for a number of insectivorous and piscivorous birds (Gilbertson et al. 1991; Kubiak et al. 1989; Tillitt et al. 1992) and is generally the most sensitive endpoint, with effects on fertility, egg production, and hatching success (Eisler 1986b). Reduced survival of offspring and growth effects in offspring through the F2 generation have also been demonstrated (American kestrel studies by Fernie et al. 2003a,b,c).

Chickens and other gallinaceous birds (e.g., pheasant) are among the most sensitive species tested for effects of PCBs and dioxins. Among studies with non-gallinaceous birds, the passerine northern bobwhite (*Colinus virginianus*) appears to be less sensitive to effects of ingested Aroclors (Heath et al. 1969; Scott 1977). Bird embryos are the most sensitive life stage for assessing the effects of contaminants (Elliott et al. 1996; Kubiak and Best 1991).

Avian TRVs for dietary exposure were developed using Aroclor 1254 or "environmental PCBs" that might be representative of the exposure pathways that could occur at the Site. Only those studies that were conducted over at least a 2-month period were included in this assessment. This was done because there are large number of LD_{50} toxicological studies or short-term (e.g., single dose) studies that are not relevant to environmental exposures. The studies included in the dietary-TRV derivation are summarized briefly below.

Peakall (1971) exposed ring doves (*Streptopelia risoria*) to a diet containing 10 ppm of Aroclor 1254 for 6 months and evaluated whether there was any impact on eggshell thickness (based on washed eggshell weights) relative to a control diet. The eggshell weights of exposed and control birds were comparable. The 10 ppm represents a NOAEL. The author did not include the body weight or ingestion rates of the test organisms. Using the average body weight (0.155 kg) and ingestion rate (0.017 kg/day) reported by Sample et al.

(1996), this equates to a NOAEL of 1.1 mg/kg-day. A LOAEL could not be calculated from this study.

Dahlgren et al. (1972) exposed ring-necked pheasants (*Phasianus colchicus*) for 17 weeks via oral gavage and monitored reproduction. Two dose levels were used (12.5 and 50 mg/kg of Aroclor 1254). USEPA (2000b) considered the lower dose to be a NOAEL and the upper dose the LOAEL. Adjusting to a daily dose using the average body weights of the test organisms, yielded a NOAEL and LOAEL of 1.8 and 7.1 mg/kg-day, respectively.

Heath et al. (1972) evaluated the toxicity of Aroclor 1254 in the northern bobwhite (*Colinus virginianus*) and Mallard duck (*Anas platyrhynchos*). Over a 2-year period, birds were fed diets containing either 25 or 50 ppm of Aroclor 1254 and were evaluated for egg production, egg hatchability, and survival of chicks. The NOAEL for the northern bobwhite was 50 ppm in the diet, while the NOAEL for the duck was 25 ppm (and the LOAEL was 50 ppm). Adjusting to a daily dose using the average body weights and ingestion rates of the test organisms resulted in NOAEL values of 4.7 and 7 mg/kg-day for the northern bobwhite and duck, respectively, and a LOAEL of 14 mg/kg-day for the duck.

Platonow and Reinhart (1973) evaluated the toxicity of Aroclor 1254 in the chicken (*Gallus domesticus*). Birds were fed diets containing either 5 or 50 ppm of Aroclor 1254 for 39 weeks. The 50 ppm dose significantly reduced production of eggs and hatchability, and was replaced with the control ration after 14 weeks. The 5 ppm level of PCB reduced egg production but not hatchability of fertile eggs. Fertility of eggs in the 5 ppm group also declined after 14 weeks of exposure, but the authors reported that this was not related to PCB exposures. Therefore, the 5 ppm level represented a LOAEL. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used to develop TRVs. The 5 ppm dose level resulted in a LOAEL of 0.35 mg/kg-day. A NOAEL could not be calculated from this study.

Peakall and Peakall (1973) evaluated the second-generation ring doves from their prior study that exposed the first-generation group to a diet containing 10 ppm of Aroclor 1254 (Peakall 1971). The second generation doves were inconsistent in incubation of their eggs resulting in reduced hatchability. The dietary value of 10 ppm was considered a LOAEL, which

yielded a LOAEL of 1.1 mg/kg-day using the same assumptions for ingestion rate and body weight as used for evaluating the Peakall (1971) study. A NOAEL could not be calculated from this study.

Cecil et al. (1974) evaluated the toxicity of Aroclors 1242, 1248, and 1254 in the chicken (*G. domesticus*). For 9 weeks, birds were fed diets containing either 2 or 20 ppm of the mixed PCBs. Hatchability declined 2 weeks after hens were given the upper dose but there was no effect at the lower dose. Therefore, the 2 ppm dose represented the NOAEL and the 20 ppm dose represented the LOAEL. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used to derive TRVs. The calculated NOAEL and LOAEL were 0.14 and 1.4 mg/kg-day, respectively.

Lillie et al. (1974) evaluated the toxicity of Aroclor 1254 in the chicken (*G. domesticus*). For 9 weeks birds were fed diets containing either 2 or 20 ppm of the Aroclor 1254. There were no effects to adult body weight gain, survival, egg weight, eggshell thickness, or fertility at either dose level. Egg production was significantly reduced relative to control at the 20 ppm dose level. Feed consumption of adults was also depressed at this dose level, which may have contributed to the reduced egg production. Based on these results, the 2 ppm dietary level represents the NOAEL while the 20 ppm dose level represents the LOAEL. The authors did not report body weights for the 9-week exposure period, but did report the initial average body weight (1.953 kg) which was used for the TRV calculation. The authors reported a food consumption rate of 118.5 to 124.3 g/day (mean: 121.4 g/day) for the two dose levels for the 9-week period. The mean ingestion rate was used to calculate the TRVs. The calculated NOAEL and LOAEL were 0.124 and 1.24 mg/kg-day, respectively.

Lillie et al. (1975) evaluated the toxicity of Aroclor 1254 (as well as Aroclors 1016, 1232, 1242, and 1248) in the chicken (*G. domesticus*). Birds were fed diets containing 5, 10, or 20 ppm of the Aroclor 1254 for 8 weeks. There were no adverse effects on egg production, egg weight, eggshell thickness, feed consumption, adult body weight changes, survival, or fertility during this exposure period. Based on these results, the 20 ppm dietary level represents the NOAEL. Using the average body weights and ingestion rates reported from their prior study (Lillie et al. 1974), the NOAEL was 1.24 mg/kg-day.

Riseborough and Anderson (1975) exposed mallard ducks (*Anas platyrhynchos*) to a diet containing 40 ppm of Aroclor 1254 for approximately 4 months and monitored egg production, eggshell thickness, and related endpoints. There were no differences between ducks fed a control diet or the 40 ppm diet on any of the measured endpoints. Based on these results, the 40 ppm dietary level represents a NOAEL. The authors did not report average body weights or ingestion rates. Therefore, the values reported by Sample et al. (1996) for these terms were used to calculate the NOAEL of 4.0 mg/kg-day.

Kosutsky et al. (1979) exposed chickens to a diet containing 5 ppm of the PCB mixture Delor 105, which is 54 percent chlorine by weight (similar to Aroclor 1254). Birds were fed this diet for 6 weeks followed by 3 weeks of control diet. There were no differences relative to controls for egg production, egg weight or eggshell strength and weight. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used for these terms to estimate a TRV. The calculated NOAEL was 0.35 mg/kg-day.

Roberts et al. (1978) reported a study where ring-necked pheasants (*P. colchicus*) were exposed to Aroclor 1254 at a dietary concentration of 50 ppm. In its review of this study, USEPA (2000b) reported that there was a reduction in female fertility at this dose level, and a LOAEL of 2.9 mg/kg-day was calculated.

Custer and Heinz (1980) exposed ducks to diets containing 25 ppm of Aroclor 1254 for 1 month. There was no apparent effect on reproductive success during this period. Although a NOAEL (7.0 mg/kg-day) could be calculated from this study, it did not meet the minimum of 2 months exposure used to derive TRVs for this project.

Summer et al. (1996) exposed white leghorn chickens *(G. domesticus*) to diets containing carp collected from Saginaw Bay, Lake Huron, Michigan. The diets contained 0, 3.5, or 34 carp, which yielded total PCB concentrations in the diets of 0.3, 0.8, and 6.6 mg/kg (respectively). The chickens were fed for an 8-week period, which overlapped egg-laying. Food consumption rates were similar across the dose groups and exposure periods. The mean body weights decreased with increasing dose and exposure periods, although the authors did not evaluate whether these were statistically significant. On average, the daily egg

production and egg weights were greater with the diets containing carp relative to control. These results would suggest a potential NOAEL for growth only at the intermediate dose level. Based on the average body weight for the intermediate dose group (1.593 kg), and their average food consumption rate (91.19 g/day), the calculated NOAEL is 0.046 mg/kg-day. This value was excluded from calculation of the geometric mean TRV because of the potential influence of other chemicals.

Custer et al. (1998) evaluated the reproductive success of tree swallows (*Tachycineta bicolor*) exposed to environmental PCBs in the Fox River and Green Bay systems. Prey items (emergent insects were collected for chemical analysis). The authors reported that there were no effects on clutch size or egg hatchability in adults that had consumed diets containing up to 0.61 mg/kg of total PCBs. Based on this information, USEPA (2000b) reported a NOAEL of 0.55 mg/kg-day. This value was excluded from calculation of the geometric mean TRV because of the potential influence of other chemicals.

The geometric means of the NOAEL and LOAEL from these studies (excluding Custer and Heinz [1980], Summer et al. [1996], and Custer et al. [1998]) are 2 and 3 mg/kg-day, respectively (Table B-11).

2.2.4 Mammals

TRVs for total PCBs were derived for mink and other mammals based primarily on reproductive toxicity studies in the literature. Toxic responses of mammals to PCB exposure are highly species-specific, and younger mammals appear to be more susceptible to PCB effects than adults (Eisler 1986b). PCB-induced reproductive impairment has been demonstrated for mink (Bleavins et al. 1980; Heaton et al. 1995a,b; Tillitt et al. 1996; Wren 1991) and other mammals including mice, rats, rabbits, swine, and rhesus monkeys (Villeneuve et al. 1971; Golub et al. 1991). Mink are generally regarded as the most sensitive mammal to ingested PCBs. Caution is needed when interpreting studies because study designs differ widely. In particular, studies in which mink are fed contaminated fish collected from the field are confounded by the possibility that other chemicals were present in the fish used to dose the test animals.

2.2.4.1 PCB TRVs for Mammals Other Than Mink

Review of the ECOTOX database showed that there is a greater frequency of toxicity data reported for mice than rats, so the TRV review for PCBs focused on mice. The key studies were compiled and are briefly summarized below. Geometric means of the NOAELs and LOAELs were calculated from results of these studies, and results were used as the NOAEL and LOAEL for mammals used in this BERA (Table B-13).

Linzey (1987) evaluated reproductive success in wild caught and laboratory-reared white-footed mice (*Peromyscus leucopus*) exposed to 10 ppm of Aroclor 1254 in their diets. There was a statistically significant reduction in the number of surviving offspring per litter in the PCB-exposed mice wild caught, but no effect on other reproductive parameters (e.g., litter size at birth). Based on this information, the 10 ppm dose level represents a LOAEL. Using the average body weights (23.2 g) and average food consumption rate (0.127 g/g bw-day) reported by the author, the calculated LOAEL is 1.27 mg/kg-day.

Linzey (1988) evaluated the survival and growth of the second generation of mice from Linzey (1987) study. At the same dose level of 10 ppm of Aroclor 1254, the second generation of PCBs-treated offspring exhibited poor reproductive success relative to controls, and grew at a slower rate compared to controls. The same LOAEL calculated from Linzey (1987) is applicable to this current study.

Simmons and McKee (1992) fed white-footed mice (*P. leucopus*) diets containing Aroclor 1254 at four dietary levels (2.5, 25, 50, and 100 ppm) for 21 days and monitored survival. There was no effect of PCB exposure at the 2.5 ppm diet concentration and a slight effect at 25 ppm. The latter represents the LOAEL and the 2.5 ppm level a NOAEL. Based on the average body weight and ingestion rate, this yields NOAEL and LOAEL values of 0.36 and 3.6 mg/kg-day, respectively. The NOAEL corresponds to the "TRV-low" recommended by USEPA Region 9 (USEPA 2002).

McCoy et al. (1995) exposed three generations of old-field mice (*Permyscus polionotus*) to a diet containing 5 ppm of Aroclor 1254 for 12 months and monitored reproduction. Dietary exposure reduced the number of litters, offspring weights, and offspring survival. Sample et al. (1996) concluded that this dietary level represented a LOAEL, and based on literature

values for body weights and ingestion rates (this information was not provided by the authors), derived a LOAEL of 0.68 mg/kg bw-day.

Voltura and French (2007) fed breeding female white-footed mice (*P. leucopus*) for 4 months on diets containing a mixture of Aroclors 1242 and 1254 (ratio of 2:1) at dietary levels of 10 and 25 ppm and monitored reproductive success. There was no effect of PCB exposure on litter size at birth or weaning, although there was a statistically significant reduction in reproductive success in female mice that were fed the 25 ppm diet. Based on this information, the latter represents the LOAEL and the 10 ppm level a NOAEL. The authors calculated daily ingestion rates of 2.64 mg/kg-day for the 10 ppm diet and 6.19 mg/kg-day for the 25 ppm diet, which represents the NOAEL and LOAEL, respectively.

The geometric means of the mouse NOAELs and LOAELs from these studies are 0.98 and <2 mg/kg-day, respectively. These values are used in the BERA risk calculations for assessing PCB exposure to marsh rice rat and raccoon. Results of studies with mink were not used in calculations of hazard quotients, but provide information relevant to the uncertainty analysis and are discussed below.

2.2.4.2 PCB TRVs for Mink

Mink are not a receptor at this Site and are unusually sensitive to PCB toxicity, so the TRVs used to evaluate risk to mammals did not include TRVs for mink. The geometric mean of NOAELs reported by several studies is 0.2 mg/kg-day, and is an unbounded NOAEL value. The geometric mean of LOAELs reported by several studies is also 0.2 mg/kg-day (Table B-11). These values are not used for the BERA risk calculations for assessing PCB exposure to marsh rice rat and raccoon. Because mink may be more sensitive than rice rat and raccoon, risks to these receptors were calculated using the PCB TRVs for mammals other than mink that are discussed above.

2.3 Bis(2-ethylhexyl)phthalate

BEHP was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

2.3.1 Benthic Invertebrates

No ER-L/ER-M values or AWQC for BEHP were available for use in the evaluation of risks to benthic invertebrates. Ho et al. (1997) measured an LC_{50} of > 1,000 µg/L for BEHP in opossum shrimp and amphipods exposed for 4 days (USEPA 2012). These were the lowest LC_{50}s in the ECOTOX database. Two other studies measured higher LC_{50}s in copepods exposed for 4 days (1×10^6 µg/L and 3×10^6 µg/L). The LC_{50} from Ho et al. (1997) was divided by 10 to estimate an NOAEC of 100 µg/L in surface water. There were several NOAECs for crustaceans and one for a polychaete that were higher than 100 µg/L. This NOAEC was used as the TRV for BEHP in benthic invertebrates (Table B-14).

2.3.2 Fish

No AWQC for BEHP were available for use in the evaluation of risks to fish. ECOTOX lists a study by Heitmuller et al. (1981) that observed an LC_{50} of 550,000 µg/L in sheepshead minnows exposed for 4 days (USEPA 2012). This value was divided by an uncertainty factor of 10 to derive an NOAEC of 55,000 µg/L, which was used as the TRV for BEHP in fish (Table B-15). There were several NOAECs for fish listed by ECOTOX that were higher than this value, but there were no studies with longer exposure durations.

2.3.3 Birds

No EcoSSL is available for BEHP and it is not addressed by Sample et al. (1996). A literature review identified only one study relevant to avian toxicity. O'Shea and Stafford (1980) studied feeding rates, weight gain, and bioaccumulation of BEHP in starlings and found that wild starlings fed diets up to 260 mg/kg BEHP for 30 days did not accumulate BEHP and had higher body weights than control birds. This dietary concentration was converted to a NOAEL dose of 74.88 mg/kg-day assuming a consumption rate of 21.6 g food/day (measured in the study) and a body weight of 75 g (based on Cuthill et al. 1999). This NOAEL was selected for use in this BERA (Table B-16). No LOAEL was identified in this study.

2.3.4 Mammals

Testicular toxicity is considered a critical effect for BEHP in mammals (ATSDR 2002). ATSDR (2002) reviewed eight studies of reproductive effects in rodents fed BEHP in food for

1 to 2 years. The bounded NOAEL for reproductive effects was 5.8 mg/kg-day based on bilateral testicular aspermatogenesis in rats following 104 days of exposure; the associated LOAEL was 29 mg/kg-day (David et al. 2000, as cited in ASTDR 2002). NOAELs associated with survival and growth endpoints were higher. The NOAEL of 5.8 mg/kg-day and LOAEL of 29 mg/kg-day were selected for this BERA (Table B-10).

2.4 Carbazole as a $COPC_E$ for Benthic Invertebrates

Carbazole was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). Carbazoles are natural products of some marine organisms (Pindur and Lemster 2001) and, as such, may represent a low level of risk to marine organisms at concentrations commonly present in the marine environment. No ER-L/ER-M values, no AWQC, and no ECOTOX records were available to describe the toxicity of carbazole to marine invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

2.5 Phenol as a $COPC_E$ for Benthic Invertebrates

Phenol was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values or AWQC for phenol were available for use in the evaluation of risks to benthic invertebrates. The ECOTOX database lists a study by Kim and Chin (1995) reporting an LC_{50} of 260 µg/L for a mysid shrimp (*Archaeomysis kokuboi*) exposed for 4 days (USEPA 2012). This was the lowest LC_{50} value in the ECOTOX database for a marine invertebrate. The other LC_{50} values with exposure durations ranging up to 21 days were higher, ranging from 5,800 to 1.05×10^8 µg/L. The LC_{50} value from Kim and Chin (1995) was divided by 10 to estimate an NOAEC of 26 µg/L in surface water, which was used as the TRV for phenol for benthic invertebrates (Table B-14).

3 METALS

Metals considered $COPC_Es$ for one or more ecological receptors are aluminum, barium, cadmium, cobalt, copper, lead, manganese, mercury, nickel, thallium, vanadium, and zinc (Table B-1). Each of these $COPC_Es$ is discussed below.

3.1 Aluminum as a $COPC_E$ for Benthic Invertebrates

Aluminum was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values, no AWQC, and no ECOTOX records were available for aluminum for use in the evaluation of risks to benthic invertebrates. Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96 hours and measured an LC_{50} value of 10,000 µg/L for aluminum. This LC_{50} was divided by 10 to estimate a NOAEC of 1,000 µg/L, which was used as the TRV for aluminum in benthic invertebrates (Table B-14).

3.2 Barium as a $COPC_E$ for Benthic Invertebrates

Barium was selected as a $COPC_E$ for benthic invertebrates (Table B-1). Barium is a naturally occurring metal used in the manufacture of ceramics, pyrotechnics, paints, enamels, and television tubes, and can be released to the environment through related industrial processes and through coal and oil combustion. Barium is more soluble in sandy soils with low pH and low organic carbon content. In biota, the properties of barium allow it to replace calcium, particularly in the release of neurotransmitters and adrenal catecholamines (USEPA 2005a).

No ER-L/ER-M values, no AWQC, and no ECOTOX records were available for barium for use in the evaluation of risks to benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

3.3 Cadmium

Cadmium can be absorbed by mammals via respiration and ingestion; absorption of ingested cadmium is controlled by several factors including the age of the organism, the valence state or form ingested, and the presence of foods rich in protein or calcium (USEPA 2005b). Metal-binding, proteinaceous metallothioneins appear to protect vertebrates from

deleterious effects of high metal body burdens (Eisler 1985). Cadmium bioconcentrates, primarily in the liver and kidney (USEPA 1999a). Cadmium accumulated from water is slowly excreted, while cadmium accumulated from food is eliminated more rapidly.

Cadmium was selected as a $COPC_E$ for fish, reptiles, birds, and mammals (Table B-1).

3.3.1 Fish

Windward (2011b) performed a review of toxicity studies in fish that were fed cadmium for various periods. The study with the lowest bounded NOAEC (in food of fish) of 68 mg/kg ww was converted to 9.0 mg/kg dw based on Windward's reported moisture content estimate of 86.7 percent for the fish diet. This study also reported the lowest bounded LOAEC of 106 mg/kg ww, or 14.1 mg/kg dw (Hatakayama and Yasuo 1987, as cited in Windward 2011b) for a reduction in the number of fry produced by guppies exposed to cadmium in food for a period of 7 months. Several other unbounded NOAECs were identified by Windward (2011b). The NOAEC of 9.0 mg/kg dw in food was used as the TRV for cadmium in fish (Table B-15).

3.3.2 Birds

Birds are comparatively resistant to the toxicity of cadmium, and mallards and chickens have been reported to tolerate 200 mg/kg of cadmium in diets for protracted periods. When present at sufficiently high doses, sublethal effects of cadmium in birds are similar to those in other animals and include growth retardation, anemia, and testicular damage. To develop an EcoSSL for cadmium, USEPA reviewed 35 papers that evaluated toxicity of cadmium to birds (USEPA 2005b); these included 49 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA calculated a TRV of 1.47 mg/kg-day, based on the geometric mean of the NOAELs for growth and reproduction (USEPA 2005b). This value was selected as the NOAEL for this BERA. The NOAEL is lower than the lowest bounded LOAEL of 2.37 mg/kg-day for reproductive effects in a 12-month dietary study of chicken (USEPA 2005b), which was selected as the LOAEL for this BERA (Table B-16).

3.3.3 Mammals

Mammals are relatively resistant to the toxicity of cadmium. Absorption and retention of cadmium decrease with prolonged exposure (Eisler 1985). Cadmium absorption through ingestion is inversely proportional to intake of other metals, especially iron and calcium.

USEPA identified 145 acceptable papers containing data for cadmium toxicity to mammals (USEPA 2005b). Within these papers were 141 NOAEL and/or LOAEL results related to survival, growth, or reproduction. The geometric mean of 38 bounded NOAELs for survival, growth, or reproductive endpoints is 2 mg/kg-day. The geometric mean of the associated LOAELs is 10 mg/kg-day. These values were used as the TRVs for cadmium in mammals (Table B-10).

3.4 Chromium

Chromium was selected as a COPC$_E$ for reptiles, birds, and mammals (Table B-1).

3.4.1 Birds

USEPA (2008) reviewed the literature on avian toxicity of trivalent and hexavalent chromium and identified 13 studies with relevant data. Insufficient study results were available to derive a TRV for hexavalent chromium, but there were 18 results related to survival, growth, or reproduction in birds exposed to trivalent chromium. USEPA calculated a TRV of 2.66 mg/kg-day based on the geometric mean of the NOAELs for growth and reproduction (USEPA 2008). The lowest bounded LOAEL is 2.78 mg/kg-day, from a 180 to 190 day feeding study that found reproductive effects in black ducks (Table B-16).

3.4.2 Mammals

USEPA's (2008) review of the chromium toxicity literature identified 20 studies with data for mammalian test species; these included 16 NOAEL and/or LOAEL results for survival, growth, or reproductive endpoints in mammals exposed to trivalent chromium and 46 results for mammals exposed to hexavalent chromium. The TRVs for trivalent and hexavalent chromium, each based on the geometric mean of the NOAELs for growth and reproduction, are 2.40 and 9.24 mg/kg-day, respectively. The more conservative value of 2.40 mg/kg-day

for trivalent chromium was selected as the NOAEL for this BERA. There are no bounded LOAELs for trivalent chromium studies. The lowest unbounded LOAEL for trivalent chromium is 2.82 mg/kg-day, based on a 50-day dietary study that observed mortality in rats. The lowest bounded LOAEL for hexavalent chromium is higher. The LOAEL of 2.82 mg/kg-day for trivalent chromium was selected for this BERA (Table B-10).

3.5 Cobalt as a $COPC_E$ for Benthic Invertebrates

Cobalt was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values or AWQC are available for cobalt for use in the evaluation of risks to benthic invertebrates. The only ECOTOX record related to cobalt's effects on survival, growth, or reproduction was a report of general effects on growth at 10 µg/L in a 14-day study of Pacific oysters (Watling 1983, as cited in USEPA 2012). The result was not indicated as statistically significant. This result could not be used as the basis for a TRV for cobalt in benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96-hour and measured an LC_{50} value of 4,500 µg/L for cobalt. The LC_{50} was divided by 10 to estimate a NOAEC of 450 µg/L, which was used as the TRV for cobalt in benthic invertebrates (Table B-14).

3.6 Copper

Copper was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1). Copper occurs naturally in many animals and plants and is an essential micronutrient that animals incorporate into several enzymes. Adverse effects in vertebrates exposed to copper include hematological, hepatic, developmental, immunological, and renal impairment. Copper exerts toxic effects by binding to DNA or by generating free radicals (USEPA 1999a). Aqueous copper speciation and toxicity depend on the ionic strength of the water. Primarily it is the dissolved cupric ion (Cu^{2+}) and possibly hydroxyl complexes that are toxic to aquatic biota; copper complexes consisting of carbonates, phosphates, nitrates, ammonia, and sulfates are weakly toxic or nontoxic (USEPA 2000a). In hard waters, 43 to

88 percent of the copper is associated with suspended solids and not available to biota (Eisler 1998).

3.6.1 Benthic Invertebrates

Many aquatic species are sensitive to dissolved concentrations of copper in the range of 1 to 20 µg/L (USEPA 2000a). In aquatic invertebrates, copper causes gill damage at high concentrations, and in fishes it interferes with osmoregulation (Eisler 1998). The AWQC for copper are 3.1 µg/L (CCC) and 4.8 µg/L (CMC) for chronic and acute exposure, respectively (USEPA 2009). For copper in sediment, the ER-L (i.e., the concentration below which adverse effects rarely occur) is 34 mg/kg dw, and the ER-M (i.e., the concentration above which adverse effects are considered probable) is 270 mg/kg dw (NOAA 1999). The ER-L of 34 mg/kg and ER-M of 270 mg/kg in sediment were used as TRVs for copper in benthic invertebrates.

3.6.2 Fish

Windward (2011b) reviewed 15 toxicity studies in fish that were fed copper for various periods. All of the studies reported on growth effects; studies reporting reproductive effects were not found. The study providing the lowest bounded NOAEC and LOAEC could not be confirmed by other investigators and was not considered typical of toxicity levels in other fish species. As a result, Windward (2011b) selected the next-lowest results of a 60-day feeding study with juvenile rockfish reporting an NOAEC of 50 mg/kg and an LOAEC of 100 mg/kg dw for growth. These were used as the TRVs for copper expressed as a concentration in fish food (Table B-15).

3.6.3 Birds

Experiments with domestic poultry show that copper accumulates in livers of mallard ducklings at dietary concentrations as low as 15 mg/kg dw ration. Mehring et al. (1960) reported a NOAEL of 570 mg/kg copper and a LOAEL of 749 mg/kg dw for dietary copper exposure of chicks over a period of 10 weeks. Using standard assumptions regarding body weight (0.534 kg) and food consumption (0.044 kg/day), Sample et al. (1996) derived a NOAEL of 47 mg/kg-day and a LOAEL of 62 mg/kg-day.

USEPA (2007a) identified 107 studies with data for avian test species; these contained 205 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007a) identified a TRV of 4.05 mg/kg-day, based on the highest bounded NOAEL below the lowest bounded LOAEL, which came from an 84-day feeding study of reproductive effects in chickens. The associated LOAEL from that study is 12.1 mg/kg-day. This NOAEL and LOAEL were used for this BERA (Table B-16).

3.6.4 Mammals

Copper can be lethal to mammals at high doses (Eisler 1998). Copper is lethal in sheep when eaten for extended periods at more than 80 mg/kg diet (equivalent to 5.1 to 10.7 mg/kg-day), more than 238 mg/kg diet in pigs, and more than 4,000 mg/kg diet in rats (equivalent to more than 133 mg/kg-day). Adverse sublethal effects of copper to sensitive mammals occur at dietary levels ranging from 7.9 mg/kg-day in food to 400 mg/L in drinking water. Chronic toxicity of copper sulfate on the reproduction of mink was evaluated by Aulerich et al. (1982). Data from this study were used by Sample et al. (1996) to support development of a NOAEL of 11.7 mg/kg-day and a LOAEL of 15.1 mg/kg-day for kit mortality.

USEPA (2007a) identified 97 studies with data for mammalian test species, which contained 123 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007a) identified a TRV of 5.60 mg/kg-day, as the highest bounded NOAEL below the lowest bounded LOAEL. This result came from a 4-week feeding study in pigs in which reduced growth and mortality were observed. The associated LOAEL from the same study is 9.34 mg/kg-day. This NOAEL and LOAEL were used for this BERA (Table B-10).

3.7 Lead

Lead has no nutritional or biochemical function (NAS 1980). The mechanism by which lead acts is believed to be indirect interference in normal metal-dependent enzyme functions at specific cellular sites, but toxicity can be affected by many physical and biological variables. In controlled studies, lead adversely affects survival, growth, reproduction, development, and metabolism of most species tested (Eisler 1998). In general, organolead compounds are more toxic than inorganic lead compounds, and young, immature organisms are more susceptible to lead's effects (Eisler 1998).

Birds and mammals exhibit lead toxicity as damage to the nervous system, kidneys, liver, sterility, growth inhibition, developmental retardation, and detrimental effects in blood (Eisler 1988). Irreversible central nervous system damage and decreased intelligence at extremely low doses of lead have been observed in mammals (ATSDR 1997). Inhibition of blood δ-aminolevulinic acid dehydratase, an enzyme critical in heme formation, has been observed as a result of exposure to lead in a variety of fish, invertebrates, and birds (USEPA 2000a).

Lead was selected as a $COPC_E$ for benthic invertebrates, reptiles, birds, and mammals (Table B-1).

3.7.1 Benthic Invertebrates

In aquatic environments, dissolved lead is the most toxic form; organolead compounds are much more toxic to aquatic organisms than are inorganic lead compounds (Eisler 1988; USEPA 2000a). The common forms of dissolved lead are lead sulfate, lead chloride, lead hydroxide, and lead carbonate, but the distribution of salts is highly dependent on the pH of the water. Most lead entering surface waters precipitates in sediment as carbonates or hydroxides. Bioavailability from sediment is controlled by the sediment organic content and acid-volatile sulfide (AVS) concentration (USEPA 2000a).

Lead is accumulated by aquatic organisms equally from water and through food (USEPA 2000a). Although methylated lead is rapidly bioaccumulated from the water by trout, there is no evidence that lead biomagnifies in the aquatic environment.

The AWQC for benthic invertebrates are a CCC of 8.1 and a CMC of 210 µg/L for chronic and acute exposure, respectively (USEPA 2009). For sediment, the ER-L for lead is 46.7 mg/kg dw, and the ER-M is 218 mg/kg dw (NOAA 1999). The ERL of 46.7 mg/kg and the ER-M of 218 mg/kg in sediment were used as the TRVs for lead in benthic invertebrates.

3.7.2 Birds

A review of wildlife toxicity studies by Eisler (1988) reports that among sensitive species of birds, survival was reduced at doses of 75 to 150 mg lead(II)/kg bw or 28 mg alkyl lead/kg bw, reproduction was impaired at dietary levels of 50 mg lead(II)/kg, and signs of poisoning were evident at doses as low as 2.8 mg alkyl lead/kg bw.

USEPA (2005c) identified 54 papers containing avian toxicity data; within these there were 57 NOAEL and/or LOAEL results related to survival, growth, or reproduction. The final NOAEL of 1.63 mg/kg-day developed by USEPA (2005c) is the highest bounded NOAEL below the lowest bounded LOAEL, based on a dietary study of reproductive effects in chickens. The minimum bounded LOAEL is 1.94 mg/kg-day, based on reproductive effects observed in a 5-week feeding study in Japanese quail. Therefore, the NOAEL of 1.63 mg/kg-day and the LOAEL of 1.94 mg/kg-day were selected as the TRVs for birds for this BERA (Table B-16).

3.7.3 Mammals

Among sensitive species of mammals, survival was reduced at acute oral doses of lead as low as 5 mg/kg bw in rats, at chronic oral doses of 0.3 mg/kg bw in dogs, and at dietary levels of 1.7 mg/kg bw in horses. USEPA identified 223 individual NOAEL or LOAEL results relevant to survival, growth, or reproduction in mammals exposed to lead in toxicological studies (USEPA 2005c). USEPA derived the TRV from the highest bounded NOAEL below the lowest bounded LOAEL, which was 4.7 mg/kg-day from a 7-week drinking water study that observed growth effects in rats. The minimum bounded LOAEL is 5.0 mg/kg-day, based on reduced growth observed in a 21-day drinking water study in rats. The NOAEL of 4.7 mg/kg-day and the LOAEL of 5.0 mg/kg-day were selected as the TRVs for this BERA (Table B-10).

3.8 Manganese as a $COPC_E$ for Benthic Invertebrates

Manganese was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ ER-M values or AWQC are available for manganese for the evaluation of risks to benthic invertebrates. An ECOTOX record related to the effect of manganese on survival, growth, or reproduction was a report of no effect on growth at 10 μg/L in a 14-day study of Pacific

oysters (Watling 1983, as cited in USEPA 2012). An effect level and the percentage of the tested organisms affected were not reported, and the result was not indicated as statistically significant.

Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96 hours and measured an LC_{50} value of 70,000 µg/L for manganese. This LC_{50} was divided by 10 to estimate an NOAEC of 7,000 µg/L for manganese (Table B-14).

3.9 Mercury

Mercury is a toxic, non-essential element (NAS 1980; USEPA 1999a). Common bacteria convert inorganic forms of mercury to organic forms (Matilainen et al. 1991). Inorganic mercury is less toxic than organomercury compounds, with methylmercury being of greatest concern for potential to cause toxicity in birds and mammals. Methylmercury is highly stable and bioaccumulates and biomagnifies in food chains (USEPA 1999a).

The mechanism of mercury toxicity in animals is interference with metabolism and cell division. Mercury binds strongly with sulfhydryl groups causing inhibition or inactivation of proteins containing thiol ligands and ultimately leading to meiotic disturbances (USEPA 1999a). In all vertebrate receptors, the target organs are the kidney and central nervous system.

At low doses to birds and mammals, mercury adversely affects reproduction, growth, and development, behavior, blood and serum chemistry, motor coordination, vision, hearing, histology, and metabolism. In mammals, methylmercury irreversibly damages the central nervous system and can also be teratogenic and mutagenic. For all organisms tested, early developmental stages were the most sensitive to mercury. Numerous biological and abiotic factors modify the toxicity of mercury compounds, sometimes by an order of magnitude or more, but the mechanisms are not clear (Eisler 1987)

Mercury adversely affects reproduction, growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange in marine and freshwater organisms. Lethal

concentrations of total mercury to sensitive, representative organisms varied from 0.1 to 2.0 µg/L for aquatic fauna. Reproduction was inhibited among sensitive species of aquatic organisms at water concentrations of 0.03 to 0.1 µg/L (Eisler 1987).

Mercury was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

3.9.1 Benthic Invertebrates

The ambient water quality criteria for use in evaluation of risks to benthic invertebrates are 0.94 and 1.8 µg/L for chronic and acute exposures to mercury, respectively (USEPA 2009). For benthic invertebrates in sediment, the ER-L is 0.15 mg/kg dw, and the ER-M is 0.71 mg/kg dw (Long et al. 1995). The ER-L of 0.15 mg/kg and the ER-M of 0.71 mg/kg in sediment were used as the TRVs for mercury in benthic invertebrates.

3.9.2 Fish

Windward (2011a) reviewed six studies of mercury toxicity to fish and selected Matta et al. (2001), who exposed mummichog (*Fundulus heteroclitus)* to methylmercuric chloride in food in a multi-generational test of reproductive effects. Exposure to methylmercury reduced male survival in the parental generation, reduced the ability of offspring to successfully reproduce, and altered sex ratios in offspring. The reduced male survival could have been due to increased aggression observed in treated males but not in treated females; the aggression is consistent with neurotoxic effects of methylmercury but might also be due to aquarium confinement. Based on the findings of Matta et al. (2001), the NOAEL is 0.5 mg/kg in food and the LOAEL is 1.9 mg/kg in food (Table B-15).

3.9.3 Birds

Hill and Schaffner (1976) found a NOAEL of 4 mg/kg diet for reproductive effects in Japanese quail. The LOAEL was 8 mg/kg diet for decreased fertility and hatchability of eggs. Using the data reported in this study, Sample et al. (1996) developed a NOAEL intake of 0.45 mg/kg-day and a LOAEL intake of 0.9 mg/kg-day. Heinz (1979) administered methyl mercury dicyandiamide in the diet to 3 generations of mallard duck. This study reported a chronic LOAEL of 0.5 mg/kg diet for decreased production of eggs and ducklings in the third

and second generation, respectively, but 0.5 mg/kg diet did not result in adverse effects in the first generation. This study was not selected to support the TRV for mercury because it does not provide an appropriate model for expected environmental conditions: it is very unlikely that birds using the Site would be exposed to the same concentration over three consecutive generations. Therefore, for the purposes of this BERA, 0.5 mg/kg diet was considered to represent a NOAEL for mercury in ducks. Using the consumption rate reported by Heinz (1979) of 0.156 kg food/kg bw-day from the study, the one-generation NOAEL of 0.078 mg/kg-day was derived for this BERA. The LOAEL of 0.9 mg/kg-day for reproductive effects from the Hill and Schaffner (1976) study was selected as the LOAEL for this BERA (Table B-16).

3.9.4 Mammals

Mercury administered to test animals as inorganic salts tends to be less toxic than is organic methylmercury. Wobeser et al. (1976) administered methylmercury chloride in the diet to mink over a period of 93 days. They found a NOAEL of 1.1 mg/kg diet and a LOAEL of 1.8 mg/kg ww diet for mortality, weight loss, and behavioral abnormalities. Sample et al. (1996) used the data from this study, combined with a subchronic–chronic uncertainty factor of 0.1, to calculate a NOAEL of 0.015 mg/kg-day and a LOAEL of 0.025 mg/kg-day, which were selected as the TRVs for mammals for this BERA (Table B-10). Because of the uncertainty factor applied, the highly toxic form of mercury used, and the relatively high sensitivity of mink relative to other mammals, these TRVs are considered to be very conservative.

3.10 Nickel

Nickel was selected as a $COPC_E$ for fish, reptiles, birds, and mammals (Table B-1).

3.10.1 Fish

The AWQC for nickel (USEPA 2009) were not used as the TRV for nickel in fish because they are based largely on responses in freshwater species. At the time the criteria were derived, there were no paired acute and chronic data for marine fish (or other taxa), and the acute-to-chronic ratio used in the calculations seems to greatly overestimate toxicity to marine species (Hunt et al. 2002). An extensive study with several marine species, including

a fish, the topsmelt (*Atherinops affinis*), reports results from both acute and chronic toxicity tests. Fish growth was not affected in the chronic test, and the endpoint was mortality. The lower of two chronic effects levels for mortality in topsmelt was 3,240 µg/L. From these and the acute results, an acute-to-chronic ratio of 6.22 is presented by Hunt et al. (2002), which was in close agreement for all marine species tested in the study.

Because of the new information provided by the Hunt et al. (2002) study, the nickel TRV for fish was derived as follows. Each of the species mean acute values for marine fish provided by USEPA (1986) were divided by an uncertainty factor of 10 to estimate corresponding NOECs, and these were combined with the lowest NOEC for topsmelt from Hunt et al. (2002). From these data, a geometric mean was calculated as 3,595 µg/L. All of the data used in this calculation are presented in Table B-17. This estimated NOAEL for fish, set at two significant figures (3,600 µg/L) was used as the nickel TRV for fish (Table B-15).

3.10.2 Birds

USEPA (2007b) identified 11 studies with data for avian test species. In these studies, there were 17 NOAEL and/or LOAEL results for survival, growth, or reproductive endpoints. USEPA (2007b) derived a TRV of 6.71 mg/kg-day based on the geometric mean of the NOAEL values for growth and reproduction. The minimum bounded LOAEL is 11.5 mg/kg-day from a 42-day feeding study in chickens in which growth effects were observed. The NOAEL of 6.71 mg/kg-day and the LOAEL of 11.5 mg/kg-day were used for this BERA (Table B-4).

3.10.3 Mammals

USEPA identified 61 individual NOAEL or LOAEL results relevant to survival, growth, or reproduction in mammals exposed to nickel in toxicological studies (USEPA 2007b). USEPA derived a TRV of 1.7 mg/kg-day based on the highest bounded NOAEL below the lowest bounded LOAEL. This value came from a study in which juvenile mice dosed orally for 35 days exhibited reproductive effects. The minimum bounded LOAEL is 2.71 mg/kg-day from a 35-day study in which mice exposed orally exhibited reproductive effects. The NOAEL of 1.7 mg/kg-day and the LOAEL of 2.71 mg/kg-day were used for this BERA (Table B-10).

3.11 Thallium as a $COPC_E$ for Benthic Invertebrates

Thallium has applications in rodenticides and insecticides (banned in the U.S. since 1975), treatment of skin infections, manufacture of glass and semiconductors, and infrared detectors. It is considered highly toxic to mammals. Thallium was selected as a $COPC_E$ for benthic invertebrates (Table B-1).

Relatively little information on the toxicity of thallium to aquatic life is available. According to USEPA (1986), acute thallium toxicity to aquatic life in saltwater occurs at concentrations as low as 2,130 µg/L. According to a review by Peter and Viraraghavan (2005), thallium kills insects at 2 mg/L, tadpoles at 0.4 mg/L, and fish at 1 mg/L. There are no AWQC for thallium and no ECOTOX records related to benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles. The acute toxicity value of 2,130 µg/L was divided by 10 to estimate a NOAEC of 213 µg/L in surface water, which was used as the TRV for thallium in benthic invertebrates (Table B-14).

3.12 Vanadium

Vanadium can exist in many valence states (most often 5+) and is common in the earth's crust. It is used in ferrous metallurgy in the manufacture of special steels. Alloys of vanadium with non-ferrous metals are used in aircraft and space technology. Sources to the environment include combustion of fossil fuels and disposal of coal wastes and flyash.

Vanadium was selected as a $COPC_E$ for benthic invertebrates, reptiles, and birds (Table B-1).

3.12.1 Benthic Invertebrates

No ER-L/ER-M values or AWQC are available for vanadium for use in evaluation of risks to benthic invertebrates. There are no ECOTOX records related to survival, growth, or reproduction in benthic invertebrates. The Concise International Chemical Assessment Document for vanadium[5] reports that 50 µg/L is associated with impaired development of oyster (*Crassostrea gigas)* larvae following acute exposures of 48 hours (WHO 2001). This

[5] http://www.inchem.org/documents/cicads/cicads/cicad29.htm#_29ci1A10

was the most sensitive species for which data are available. Development of urchin (*Paracentrotus lividus*) larvae was impaired at 100 µg/L but not at 50 µg/L; and mortality was observed in brine shrimp (*Artemia salina*) after 8-day exposures at 250 µg/L. The acute effect level in oysters of 50 µg/L was divided by 5 to estimate a LOAEC of 10 µg/L, and divided by 10 to estimate a NOAEC of 5 µg/L in marine water. These were used as the TRVs for vanadium in benthic invertebrates (Table B-14).

3.12.2 Birds

Several studies report on the toxicity of vanadium to birds (USEPA 2005d), many of which address survival, growth, and reproductive endpoints. However, the majority of data are for the chicken, with only two studies reporting toxicity to ducks, and one to Japanese quail. USEPA (2005d) identified a TRV of 0.344 mg/kg-day based on the highest bounded NOAEL below the lowest bounded LOAEL for a survival, growth, or reproductive endpoint. This value came from a 5-week feeding study that observed reduced growth in chickens. The lowest bounded LOAEL is 0.413, from a 7-day feeding study in which reproductive effects were observed in chickens. The NOAEL of 0.344 mg/kg-day and the LOAEL of 0.413 mg/kg-day were used for this BERA (Table B-16).

3.13 Zinc

Zinc is required for normal growth, development, and function in all animal species that have been studied (NAS 1980). Zinc attaches to organic molecules such as amino acids, proteins, and nucleic acids, directly binding to sulfhydryl, amino, imidazole, and phosphate groups (NAS 1980). Zinc has low toxicity to birds and mammals, and is not a highly mobile element in aquatic food webs and does not biomagnify in marine or freshwater food webs (USEPA 2000a). Exposures to high concentrations of zinc may result in reduced growth, anemia, reduced bone ash, decreased tissue concentrations of iron, copper, and manganese, and decreased use of calcium and phosphorus (NAS 1980).

Zinc in the water column can partition to dissolved and particulate organic carbon. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc since the divalent zinc ion is believed to be responsible for observed biological effects (USEPA 2000a). Significant adverse effects of

zinc on survival, growth, and reproduction occur in sensitive species of aquatic plants, protozoans, sponges, molluscs, crustaceans, echinoderms, fish, and amphibians at nominal water concentrations between 10 and 25 µg/L (Eisler 1993).

Zinc was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

3.13.1 Benthic Invertebrates

Bioavailability of zinc in sediments is controlled by the AVS concentration. The ER-L for zinc is 150 mg/kg dw and the ER-M is 410 mg/kg dw (NOAA 1999). The ER-L and ER-M were used as the TRVs for zinc in benthic invertebrates.

3.13.2 Fish

In general, zinc is more toxic to fish embryos and juveniles than to adult fish. Windward (2011b) derived a dietary NOAEL of 1,900 mg/kg based on a 60-day study that observed growth and survival in rainbow trout fry (Mount et al. 1994) and a dietary LOAEL of 2,000 mg/kg based on a 6-week study that observed growth in rainbow trout fingerling (Takeda and Shimma 1977). These values were selected for this BERA (Table B-15).

3.13.3 Birds

Growth of domestic poultry and wild birds was reduced at concentrations in the diet >2,000 mg/kg, and survival was reduced at concentrations >3,000 mg/kg in diet, or at a single oral dose >742 mg/kg bw. Younger stages (i.e., chicks, ducklings) were least resistant (Eisler 1993). A study of dietary exposure of white Leghorn hens to zinc sulfate for 44 weeks found a NOAEL of 228 mg/kg diet and a LOAEL of 2,028 mg/kg diet for decreased egg hatchability. Sample et al. (1996) used data from this study to develop a NOAEL intake of 14.5 mg/kg-day and a LOAEL intake of 131 mg/kg-day.

USEPA (2007c) found 53 studies with data for avian test species. Within these studies, there were 94 NOAEL and/or LOAEL results related to survival, growth, or reproduction, all of which were for domestic poultry. USEPA (2007c) calculated a TRV of 66.1 mg/kg-day based on the geometric mean of NOAEL values for growth and reproduction, but NOAELs ranged

as high as 741 mg/kg-day (survival in turkeys), 367 mg/kg-day (reproduction in chickens), and 106 mg/kg-day (growth of chickens). The lowest bounded LOAEL for a growth endpoint was 86.6 mg/kg-day, based on a 14-day feeding study that observed growth reduction in Japanese quail. These values were selected as TRVs for birds (Table B-16).

3.13.4 Mammals

Sensitive species of livestock and small laboratory animals are adversely affected at 90 to 300 mg/kg diet, >90 mg/kg-day repeated oral dose, >300 mg/L drinking water, and >350 mg/kg bw single oral dose. A study of dietary exposure of rats to zinc oxide during gestation reported a NOAEL of 2,000 mg/kg diet and a LOAEL of 4,000 mg/kg diet for increased rates of fetal resorption and reduced fetal growth rates (Schlicker and Cox 1968). Sample et al. (1996) used data from this study to develop a NOAEL intake of 160 mg/kg-day and a LOAEL intake of 320 mg/kg-day.

USEPA's (2007c) literature review found 99 studies with data for mammalian test species. These studies contained 104 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007c) calculated a TRV of 75.4 mg/kg-day, based on the geometric mean of the NOAEL values for growth and reproduction. The lowest bounded LOAEL for a survival, growth, or reproductive endpoint is 75.9 mg/kg-day, based on a 14-week feeding study of reproductive effects in cattle. The NOAEL of 75.4 mg/kg-day and the LOAEL of 75.9 mg/kg bw-day were used in this BERA (Table B-10).

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TABLES

Table B-1
Chemicals of Potential Ecological Concern[a]

Chemical	Receptors North of I-10 and Aquatic Environment	
	Benthic Invertebrates	Fish and Wildlife
Dioxins/Furans		
Dioxins and Furans	X	X
Polychlorinated Biphenyls		
Polychlorinated Biphenyls		X
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	X	X
Carbazole	X	
Phenol	X	
Metals		
Aluminum	X	
Barium	X	
Cadmium		X
Chromium		
Cobalt	X	
Copper	X	X
Lead	X	
Manganese	X	
Mercury	X	X
Nickel		X
Thallium	X	
Vanadium	X	
Zinc	X	X

Notes

a - Some of these chemicals will be evaluated for human health risks. See Integral (2012).

Table B-2

Default Values for Body Weight and Consumption Rates for Test Species Used in Derivation of TRVs

Test Species	Body Weight (kg)	Consumption Rate (kg dw/day)
Mammals		
Mink	1.0 [a]	0.137 [a]
Mouse	0.03 [a]	NA
Rat	0.35 [a]	0.028 [a]
Birds		
Japanese quail	0.15 [a]	0.0169 [a]
Mallard duck	1.13 [b]	NA
Starling	NA	NA
Turkey	NA	0.174 [a]

Notes

NA = not available

a - Sample et al. (1996)

b - USEPA (1993)

Table B-3
Ambient Water Quality Criteria for the Protection of Aquatic Life

Chemical	Water Concentration[a] (µg/L)	
Organic Compounds		
PCBs[b]	CMC	NA
	CCC	0.03
Metals		
Cadmium[b]	CMC	40
	CCC	8.8
Copper	CMC	4.8
	CCC	3.1
Lead	CMC	210
	CCC	8.1
Mercury	CMC	1.8
	CCC	0.94
Nickel	CMC	74
	CCC	8.2
Zinc	CMC	90
	CCC	81

Notes

CCC = criterion continuous concentration: an estimate of the highest concentration in ambient water to which an aquatic community can be exposed indefinitely without an adverse effect.
CMC = criterion maximum concentration: an estimate of the highest concentration in ambient water to which an aquatic community can be exposed briefly without an adverse effect.

NA = not available

PCB = polychlorinated biphenyl

a - National ambient water quality criteria (AWQC) for the protection of aquatic life. Values for saltwater are shown. These AWQC values represent the dissolved concentration.

b - Chemical is a chemical of potential ecological concern for fish only.

Table B-4
Summary of Information on the Toxicity of 2,3,7,8-TCDD[a] to Benthic Invertebrates

Exposure Medium	Test Organism	Taxonomic Classification	Dose Administration	Exposure Duration	NOAEC/LOAEC	Units	Endpoint	Notes	Reference
Sediment	*Ampelisca abdita*	Crustacea, Amphipoda	Spiked sediment	10 days	25,000/NA	ng/kg dw sediment	Growth and mortality		Barber et al. (1998)
	Nereis virens	Annelida, Polychaeta	Field-collected sediment	180 days	656/NA	ng/kg dw sediment	Mortality	Potential co-contamination with 2,3,7,8-TCDF and PCBs noted.	Pruell et al. (1993)
					422/NA	ng/kg dw tissue			
	Macoma nasuta	Mollusca, Bivalvia	Field-collected sediment	120 days	656/NA	ng/kg dw sediment	Mortality	Presence of 2,3,7,8-TCDF and PCBs in both contaminated sediments and in study organisms noted.	Rubenstein et al. (1990)
					142/NA	ng/kg dw tissue			
	Palaemonetes pugio	Crustacea, Caridea	Field-collected sediment	28 days	656/NA	ng/kg dw sediment	Mortality		
					138/NA	ng/kg dw tissue			
	Chironomus riparius	Arthropoda, Diptera	Spiked sediment	28 days	10,000/NA	ng/kg dw sediment	Mortality, growth, mentum deformities		Loonen et al. (1996)
					14,000/NA	ng/kg dw tissue			
Water	*Daphnia magna*	Crustacea, Cladocera	Laboratory water	48 hours followed by 7 day recovery	1,030/NA	ng/kg ww tissue[b]	General toxicity		Adams et al. (1986)
	Mya arenaria	Mollusca, Bivalvia	Laboratory water	Single pulse dose for 24 hours followed by 28 day observation period	200/NA	ng/L in water	Reduced body mass over time		Cooper and Wintermyer (2009)
					NA/4.8 - 20	ng/kg ww weight tissue[b]	Gonadal lesions (female)		
	Physa sp.	Mollusca, Gastropoda	Well water	36 days followed by recovery period	200/NA	ng/L in water	Parental mortality, hatching, juvenile mortality		Miller et al. (1973)
	Paranais sp.	Annelida, Oligochaeta	Well water	55 days	200/NA	ng/L in water	Total biomass		
	Aedes aegypti	Arthropoda, Diptera	Well water	17 days followed by recovery period	200/NA	ng/L in water	Pupation		
	Mya arenaria	Mollusca, Bivalvia	Sea water	24 hours followed by recovery period	2,000/NA	ng/L in water	Mortality, shell length, gonadal histopathology	Tissue concentrations were measured but were widely variable among organs.	Rhodes et al. (1997)
	Helisoma sp.	Mollusca, Gastropoda	Spiked soil flooded with water	32 days	4.2/NA	ng/L in water	Reproductive activity, feeding, growth		Yockim et al. (1978)
	Daphnia magna	Crustacea, Cladocera	Spiked soil flooded with water	32 days	4.2/NA	ng/L in water	Reproductive activity, feeding, growth		
	Physa sp.	Mollusca, Gastropoda	Water		1,300/NA	ng/L in water	Reproductive activity, growth, feeding		Isensee and Jones (1975)
	Daphnia magna	Crustacea, Cladocera	Water		1,300/NA	ng/L in water	Reproductive activity, growth, feeding		

Table B-4
Summary of Information on the Toxicity of 2,3,7,8-TCDD[a] to Benthic Invertebrates

Exposure Medium	Test Organism	Taxonomic Classification	Dose Administration	Exposure Duration	NOAEC/LOAEC	Units	Endpoint	Notes	Reference
Diet	*Chironomus dilutus*	Arthropoda, Diptera	Spiked diet	35 days	3,804/NA	µg/kg TOC diet	Mortality, growth emergence, eggs/female, hatchability	TCDD concentrations also given as dw of food (323 µg/kg dw diet) and also provided on a lipid basis in *Chironomus* , see below.	West et al. (1997)
	Lumbriculus variegatus	Annelida, Oligochaeta	Spiked diet	28 days	3,594/NA	µg/kg TOC diet	Number of organisms, total biomass	Trend for reduced number of animals, but not statistically significant. TCDD concentration also given as dw of food (1,319 µg/kg dw diet) and also provided on a lipid basis in *Lumbriculus* tissue, see below.	
	Chironomus dilutus	Arthropoda, Diptera	Spiked diet	35 days	5,084	µg/kg lipid	Mortality, growth emergence, eggs/female, hatchability	Highest concentration (average of exposure group) during exposure period, achieved at day 13.	West et al. (1997)
	Lumbriculus variegatus	Annelida, Oligochaeta	Spiked diet	28 days	9,533/NA	µg/kg lipid	Number of organisms, total biomass	Highest concentration (average of exposure group), achieved at end of exposure period.	
Administered/ Injection	*Mya arenaria*	Mollusca, Bivalvia	Injection (muscle; single dose)		200/NA	ng/kg ww tissue[b]	Reduced body mass over time		Cooper and Wintermyer (2009)
			Siphon gavage (single dose)		NA/200	ng/kg ww tissue[b]	Reduced body mass over time		
	Crassostrea virginica	Mollusca, Bivalvia	Injection (Days 1 and 14)	28 day observation period	NA/2.0	ng/kg ww tissueb[a]	Reduced body mass over time		Cooper and Wintermyer (2009)
					NA/2.0	ng/kg ww tissue[b]	Gonadal lesions		
					NA/2	ng/kg ww tissue[b]	Reduced larval survival		
	Crassostrea virginica	Mollusca, Bivalvia	Injection (Days 1 and 14)	28 day observation period	NA/2	ng/kg ww tissue[b]	Delayed gonadogenesis (females)	Marked effect of solvent on this endpoint.	Wintermyer and Cooper (2007)
					NA/10	ng/kg ww tissue[b]	Sex ratio (reduced females)		
					NA/2	ng/kg ww tissue[b]	Reduced vitellogenic oocytes (females; electron microscopy)	Other reproductive endpoints also affected this exposure level.	
					2/10	ng/kg ww tissue[b]	Delayed gonadogenesis (males)		

Notes

Animals exposed to field-collected sediment may have been exposed to mixtures.
LOAEC = lowest-observed-adverse-effects concentration
NA = not available
NOAEC = no-observed-adverse effects concentration
PCB = polychlorinated biphenyl
TCDD = tetrachlorodibenzo-*p* -dioxin
TOC = total organic carbon
a - All laboratory studies summarized here used 2,3,7,8-TCDD as the exposure chemical. Some field studies summarized also measured other organocontaminants and these have been summarized in the Notes column.
b - Soft body tissue only (excluding shell)

Table B-5
Geometric Means of No-Observed-Effect Levels and Lowest-Observed-Effect Levels of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin (2,3,7,8-TCDD) and Dioxin-Like Compounds in Fish Eggs or Embryos

Common Name	Scientific Name	GM [a] (ng/g lipid)	Reference
Brook trout	*Salvelinus fontinalis*	1.68	b
Channel catfish	*Ictalurus punctatus*	11.95	c
Fathead minnow	*Pimephales promelas*	13.32	c
Japanese medaka	*Oryzias latipes*	22.66	c
Lake herring	*Coregonus artedii*	3.29	c
Lake trout	*S. namaycush*	0.42	d
Northern pike	*Esox lucius*	34.85	c
Rainbow trout	*O. mykiss*	3.05	e
White sucker	*Catastomus commersoni*	40.69	c
Zebra fish	*Danio danio*	54.17	c

Notes

GM = geometric mean

a - Values are listed with the number of significant figures presented in the respective study.

b - Johnson et al. (1998), as cited in Steevens et al. (2005)

c - Elonen et al. (1998), as cited in Steevens et al. (2005)

d - Walker et al. (1994), as cited in Steevens et al. (2005)

e - Walker and Peterson (1991), as cited in Steevens et al. (2005)

Table B-6
Summary of Egg Mortality TRVs; Maternal Transfer and Yolk Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	328	1,477	MT	a	Egg concentrations estimated on the basis of maternal dose of 1 µg/kg for no effects and an estimated 50 percent egg mortality at 4.5 µg/kg bw, assuming a 1 percent maternal transfer into eggs (mean egg wt = 30.5 g) Nosek et al. (1992a; 1993).
$[Egg]_{TCDD}$	100	1,000	YI	b	Egg concentration associated with 10 percent egg mortality
Geometric mean for pheasants	181	1,215			
Double crested cormorant					
$[Egg]_{TCDD}$	1,000	4,000	YI	c	LOAEL is associated with 23.3 percent increase in egg mortality over egg mortality in vehicle controls
$[Egg]_{TCDD}$	1,300	5,400	YI	d	LOAEL is associated with 25.5 percent increase in egg mortality over egg mortality in vehicle controls
Geometric mean for cormorants	1,140	4,648			
Final geometric mean	450	2,400			Geometric means rounded to 2 significant figures for use as TRVs
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	YI	e	LOAEL is associated with 100 percent egg mortality over control egg mortality
$[Egg]_{TCDD}$	80	160	YI	f	LOAEL is associated with 63.8 percent increase in egg mortality over egg mortality in vehicle controls
Geometric mean for chickens	89	220			
Geometric mean - all	260	1,100			

Notes

LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
MT = maternal transfer
NOAEC = no-observed-adverse effects concentration
TRV = toxicity reference value
YI = yolk injection

a - Nosek et al. (1992b)
b - Nosek et al. (1993)
c - Powell et al (1997a)
d - Powell et al. (1998)
e - Henshel et al. (1997a)
f - Powell et al. (1996)

Table B-7
Toxicity Equivalency Factors for Dioxins and Furans and Dioxin-Like PCBs

Compound	TEF-M (WHO 2005) [a]	TEF-Fish (WHO 1998)	TEF-Bird (WHO 1998)
Chlorinated Dibenzo-*p*-Dioxins			
2,3,7,8-TCDD	1	1	1
1,2,3,7,8-PeCDD	1	1	1
1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
1,2,3,4,6,7,8-HpCDD	0.01	0.001	0.001
OCDD	0.0003	0.0001	0.0001
Chlorinated Dibenzofurans			
2,3,7,8-TCDF	0.1	0.05	1
1,2,3,7,8-PeCDF	0.03	0.05	0.1
2,3,4,7,8-PeCDF	0.3	0.5	1
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
OCDF	0.0003	0.0001	0.0001
Non-ortho Substituted PCBs			
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.0001	0.0001	0.05
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.0003	0.0005	0.1
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	0.005	0.1
3,3',4,4',5,5'-Hexachlorobiphenyl (PCB 169)	0.03	0.00005	0.001
Mono-ortho Substituted PCBs			
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.00003	0.000005	0.0001
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.00003	0.000005	0.0001
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00003	0.000005	0.00001
2',3,4,4',5-Pentachlorobiphenyl (PCB 123)	0.00003	0.000005	0.00001
2,3,3',4,4',5-Hexachlorobiphenyl (PCB 156)	0.00003	0.000005	0.0001
2,3,3',4,4',5'-Hexachlorobiphenyl (PCB 157)	0.00003	0.000005	0.0001
2,3',4,4',5,5'-Hexachlorobiphenyl (PCB 167)	0.00003	0.000005	0.00001
2,3,3',4,4',5,5'-Heptachlorobiphenyl (PCB 189)	0.00003	0.000005	0.00001

Sources

WHO (1998) corresponds to Van den Berg et al. (1998)

WHO (2005) corresponds to Van den Berg et al. (2006)

Notes

PCB = polychlorinated biphenyl

TEF-M = mammalian toxicity equivalency factor

a - Endorsed by USEPA (2010)

Table B-8
Summary of Egg Mortality TRVs for TCDD; Air Cell or Albumin Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Japanese quail					
$[Egg]_{TCDD}$	3,540	9,020	CI	a	Egg mortality significantly elevated over control
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	264	1,030	CI	a	Egg mortality significantly elevated over control
$[Egg]_{TCDD}$	100	1,000	AI	b	LOAEL is egg concentration associated with 20 percent mortality over control
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	CI	c	LOAEL is egg concentration associated with 35 percent increase in mortality over control
$[Egg]_{TCDD}$	248	515	CI	a	Egg mortality significantly elevated over control

Notes

AI = albumin injection
CI = air cell injection
LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
NOAEC = no-observed-adverse effects concentration
TCDD = tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
a - Cohen-Barnhouse et al. (2011)
b - Nosek et al. (1993)
c - Henshel et al. (1997a)

Table B-9
Summary of NOAECs for Bird Eggs, Field Studies

Receptors	TEQ (ng/kg ww)	Location	Ref.	Comments
Spotted sandpipers	732	Hudson River, NY	Custer and Custer (2010)	NOAEC is based on the geometric mean of TEQ concentrations measured for the Hudson River site
Great blue heron	207	British Columbia, Canada	Elliott et al. (2001)	Based on lack of gross abnormalities, number of fledglings and hatching success at the Nicomekl River site.
Cormorants, herons and egrets	452	Galveston Bay, Tx	Frank et al. (2001)	TEQ ranged from 67 to 452 ng/kg ww, no deformities or abnormalities were detected over this range.
Osprey	136	Castle Rock and Petenwell Flowages site, WI	Woodford et al. (1998)	Using the reproductive endpoints of egg hatching and chick fledgling rates

Notes
NOAEC = no-observed-adverse effects concentration

Table B-10
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	0.98	a	Reproduction	Geometric means of NOAELs and LOAELs from toxicity studies with mice.
	LOAEL	2			
TCDD	NOAEL	0.000001	b	Reproduction	Converted from dietary concentration to dose using assumed body weight and consumption rate.
	LOAEL	0.00001			
Bis(2-ethylhexyl)phthalate	NOAEL	5.8	c	Reproduction	Effects seen at 29 and 147 mg/kg/day doses might be age-related, in which case NOAEL and LOAEL would be under-estimated
	LOAEL	29			
Metals					
Cadmium	NOAEL	2	d	Geometric mean of bounded NOAELs for growth, mortality, repro	38 bounded NOAELs/LOAELs included in calculation
	LOAEL	10		Geometric mean of associated LOAELs	
Chromium	NOAEL	2.40	e	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.82		Mortality	No unbounded LOAELs. This is the minimum unbounded LOAEL for a mortality/growth/repro endpoint.
Copper	NOAEL	5.6	f	Reproduction, growth, survival	Highest bounded NOAEL beneath the lowest bounded LOAEL
	LOAEL	9.34			
Lead	NOAEL	4.7	g	Survival	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	5.0		Growth	Lowest bounded LOAEL
Mercury	NOAEL	0.015	h	Survival and growth	Converted from dietary concentration to dose using assumed body weight and consumption rate. Converted to chronic from subchronic exposure period. Administered as methylmercury chloride.
	LOAEL	0.025			
Nickel	NOAEL	1.7	i	Reproduction	Highest bounded NOAEL below the lowest bounded LOAEL for a mortality/growth/repro endpoint
	LOAEL	2.71			Minimum bounded LOAEL for a mortality/growth/repro endpoint

Table B-10
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Zinc	NOAEL	75.4	j	Reproduction	Geometric mean of NOAELs for reproduction and growth; lowest bounded LOAEL for survival, reproduction and growth
	LOAEL	75.9			

Notes

EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p*-dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Aulerich and Ringer (1977)
b - Murray et al. (1979)
c - David et al. (2000)
d - EcoSSL (USEPA 2005b)
e - EcoSSL (USEPA 2008)
f - EcoSSL (USEPA 2007a)
g - EcoSSL (USEPA 2005c)
h - Sample et al. (1996)
j - USEPA (2007c)

Table B-11
Summary of Toxicity Reference Values for Total PCB Effects on Fish, Birds, and Mammals

Receptors	TRV_{NOAEL} GM (range)	TRV_{LOAEL} GM (range)	Units	TRV Basis	Sample Size
Fish	5.0 (1.9–15)	16 (2.7–170)	mg/kg ww	Residue in whole fish	NOAEL n=3 across 3 fish species; LOAEL n=3 across 3 fish species
Birds	≥2 (0.1–7)	≤3 (1–10)	mg/kg-day	Ingested dose	NOAEL n=9 across 5 bird species; LOAEL n=7 across 4 bird species
Mink	≥0.2 (0.1–0.1)	≤0.2 (0.7–0.7)	mg/kg-day	Ingested dose	Mink toxicity studies; NOAEL n=4; LOAEL n=7
Other Mammals	0.98 (0.36–2.6)	≤2 (4–6)	mg/kg-day	Ingested dose	Mouse toxicity studies; NOAEL n=2; LOAEL n=5

Notes

GM = geometric mean. A ">" or "<" sign is used where one or more of the values used to determine the geometric mean TRV was an unbounded NOAEL or LOAEL, respectively. Range is the range of bounded TRVs (i.e., across accepted studies in which both a NOAEL and a LOAEL were reported). Sample size is all bounded and unbounded TRVs from accepted studies before combining studies within species.

LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
TRV = toxicity reference value
ww = wet weight

Table B-12
Data Sources Used to Develop Fish Tissue-Based Toxicity Reference Values for Total PCBs

Chemical	Chemical Form	Test Organism	Study Duration	Endpoints	Exposure Route	Dosage/Exposure	Reported Toxicity	Test Species NOAEL	Test Species LOAEL	TRV_{NOAEL} (mg/kg)	TRV_{LOAEL} (mg/kg)	Value Used for ERA	Source Category	Source/Comments
Total PCBs	Aroclor-1254	Sheephead Minnow	28 days	F1 Generation Survival	Aqueous	Five water conc (0.1, 0.32, 1.0, 3.2, and 10 µg/L) plus control	NOAEL, LOAEL	0.1 µg/L (1.9 mg/kg in adult)	0.32 µg/L (9.3 mg/kg in adult)	1.9	9.3	Yes	Lit	Hansen et al. (1973); NOAEL and LOAEL were used for the Onondaga Lake BERA (NYSDEC 2002) and Hudson River Revised BERA (USEPA 2000)
Total PCBs	Clophen A50	Common Minnow	40 days exposed, monitor for additional 260 days	Growth, F1 Generation Survival	Diet	Control and 3 dietary conc (20, 200 and 2,000 mg/kg)	NOAEL, LOAEL	15 mg/kg	170 mg/kg	15	170	Yes	Lit	Bengtsson (1980)
Total PCBs	20 PCBS (representing the 154 tetra- to hepta-chlorinated congeners)	Zebrafish	13 weeks exposed; reproduction study initiated at 9 weeks	Growth, Survival, Reproduction	Diet	Control and 3 dietary conc (0.008, 0.08, and 0.4 mg/kg)	LOAEL	NR	2.7 mg/kg	---	2.7	Yes	Lit	Orn et al. (1998). Fish were dissected and livers and ovaries removed prior to measuring concentrations. LOAEL value therefore is biased low.
Total PCBs	Aroclor-1016, Aroclor-1254, Aroclor-1260	Striped Bass	30 days (10 days yolk absorption, 20 days exposed)	Growth, Survival	Diet	Control and 1 dietary conc (0.014 and 0.127 mg/kg, respectively)	NOAEL	3.1 mg/kg	NR	4.4	---	Yes	Lit	Westin et al. (1983). No effects observed in dosed fish. Highest concentration in whole larvae selected as TRV.

Notes

TRV values are on a whole body wet weight basis.
When multiple NOAEL or LOAEL values are reported based on the same endpoint, the highest reported value was used for the TRV calculations.
Study durations shown were as reported by the authors and were also adjusted to days to facilitate comparisons between studies. Where an unbounded LOAEL was greater than the maximum bounded LOAEL, it was excluded from calculation of the geometric mean LOAEL.
Where an unbounded NOAEL was less than the minimum bounded NOAEL, it was excluded from calculation of the geometric mean NOAEL.

BERA = baseline ecological risk assessment
ERA = ecological risk assessment
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
NR = not reported or not required
PCB = polychlorinated biphenyl
TRV = toxicity reference value
USACE = U.S. Army Corps of Engineers

Table B-13
Data Sources Used to Develop Total PCBs Toxicity Reference Values for Mice

Chemical	Chemical Form	Study Duration	Endpoints	Exposure Route	Dosage	Reported Toxicity Value(s)	NOAEL	LOAEL	Calculated TRV_{NOAEL} (mg/kg-day)	Calculated TRV_{LOAEL} (mg/kg-day)	Source/Comments
Total PCBs	Aroclor 1254	9 to 18 months (270 to 540 days)	Repro	Oral in diet	10 ppm in diet	LOAEL	---	10 mg/kg	---	1.27	Linzey (1987)
Total PCBs	Aroclor 1254	F1 generation 4 to 12 weeks (28 to 84 days)	Repro, Survival	Oral in diet	10 ppm in diet	LOAEL	---	10 mg/kg	---	1.27	Linzey (1988) [a]
Total PCBs	Aroclor 1254	21 days	Survival	Oral in diet	Four dose levels: 2.5, 25, 50 and 100 ppm	NOAEL, LOAEL	2.5 ppm	25 ppm in diet	0.36	3.6	Simmons and McKee (1992) TRV_{NOAEL} value reported by USEPA (2002).
Total PCBs	Aroclor 1254	3 generations (1 year) (365 days)	Repro	Oral in diet	5 ppm in diet	LOAEL	---	5 mg/kg	---	0.68	McCoy et al (1995)
Total PCBs	Aroclors 1242 and 1254	4 months (120 days)	Growth, Repro,	Oral in diet	Two dose levels: 10 and 25 ppm as total PCBs	NOAEL, LOAEL	10 ppm in diet	25 ppm in diet	2.64	6.19	Voltura and French (2007) PCB mixture was 2:1 Aroclor 1242:Aroclor 1254.
							Accepted Studies	*Range*	0.36 - 2.64	0.68 - 6.19	
								Range of bounded values	0.36 - 2.64	3.60 - 6.19	
								Geometric Mean	0.98	2	

Notes

All mouse toxicity studies shown in this table were accepted for calculation of TRVs.
The geometric means of the TRVs were body-weight scaled to derive TRVs for mammals other than mink. TRVs specific for mink were available.
Study durations shown were as reported by the authors and were also adjusted to days to facilitate comparisons between studies.

LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
PCB = polychlorinated biphenyl
TRV = toxicity reference value

a - FI generation were offspring of parents from prior study (Linzey 1987).

Table B-14
Toxicity Reference Values and Benchmarks for Benthic Macroinvertebrates

Chemical	Sediment Concentration (ng/kg dw for organics; mg/kg dw for metals)		Ref	Water Concentration[a] (µg/L)		Ref	Endpoint/Comments
	TRV Type	Value		TRV Type	Value		
Organic Compounds							
2,3,7,8-TCDD	NOAEC	2,343		NA	NA		Geometric mean of NOAECs for a range of invertebrate taxa from Table B-4
Bis(2-ethylhexyl)phthalate	--	ND		NOAEC[b]	100	c	Opossum shrimp and amphipod mortality in 4 day lab test. NOAEC is LC_{50} ÷10.
Carbazole	--	ND					No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Phenol	--	ND		NOAEC[b]	26	d	Mysid shrimp mortality in 4 day lab test. NOAEC is LC_{50} ÷ 10.
Metals							
Aluminum	--	ND		NOAEC[b]	1,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Barium	--	ND		--	ND		No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Cobalt	--	ND		NOAEC[b]	450	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Copper	ER-L	34	f	--			
	ER-M	270	f	AWQC (CCC)	3.1	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]
Lead	ER-L	46.7	f	--			
	ER-M	218	f				
Manganese	--	ND		NOAEC[b]	7,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Mercury	ER-L	0.15	f	--			
	ER-M	0.71	f	AWQC (CCC)	0.94	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]
Thallium	--	ND		NOAEC[b]	213	h	Derived from acute toxicity to marine life . NOAEC is EC ÷ 10. Details unavailable.
Vanadium	--	ND		NOAEC	5	i	NOAEC is EC_{50} ÷10 in most sensitive species. Effect is development.
				LOAEC	10	i	LOAEC is EC_{50} ÷ 10 in most sensitive species. Effect is development.
Zinc	ER-L	150	f	--			
	ER-M	410	f	AWQC (CCC)	81	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]

Notes

-- = Risks were not evaluated using lines of evidence requiring this information
AWQC = Ambient Water Quality Criteria. Criterion Continuous Concentrations shown
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
EC = effects concentration
ER-L = effect range-low: concentration below which effects are rarely observed or predicted among sensitive life stages and (or) species of biota
ER-M = effect range-median: concentration above which effects are frequently or always observed among most species of biota
USEPA = U.S. Environmental Protection Agency
WHO = World Health Organization
a - TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - TRV is an LC_{50} divided by an uncertainty factor of 10.
c - Ho et al. (1997)
d - Kim and Chin (1995)
e - Bengtsson (1978)
f - Long et al. (1995)
g - Ambient Water Quality Criteria Website (http://water.epa.gov/scitech/swguidance/standards/current/index.cfm#altable)
h - USEPA (1986)
i - WHO (2001)

Table B-15
Toxicity Reference Values and Benchmarks for Fish

Chemical	Water Concentration[a] µg/L		Ref	Fish Food[b] (mg/kg dw)		Ref	Fish Whole Body		Units	Ref	Comments
Organic Compounds											
TCDD (mg/kg lipid)	--	--		--	--		NOAEL	0.321	µg/kg lipid	c	From a species sensitivity distribution; protects 95 percent of fish species. Endpoint is egg survival.
PCBs	--	--		--	--		NOAEL	5.0	mg/kg ww	d	Geometric mean of NOAELs from 3 fish species.
	--	--		--	--		LOAEL	16	mg/kg ww	d	Geometric mean of LOAELs across 3 fish species.
Bis(2-ethylhexyl)phthalate	NOAEL	55,000	e	--	--		--	--	--		Derived from 4-day acute test with sheepshead minnow. NOAEL is $LC_{50} \div 10$. Endpoint is survival.
Metals											
Cadmium				LOAEL	14.1	f	--	--	--		
Copper				NOAEL	50	g	--	--	--		
				LOAEL	100	h	--	--	--		
Mercury				NOAEL	0.5	i	--	--	--		Endpoint is F_0 male survival in mummichog resulting from increased aggression due to neurotoxic effects. aquarium confinement, or both.
				LOAEL	1.9	i	--	--	--		
Nickel	NOAEL	3,600	j, k	ND			--	--	--		Geometric mean of NOAECs for several marine fish. See Table B-16 and Appendix B text.
Zinc				NOAEL	1,900	l	--	--	--		Fish exposed to multiple metals in water as well as food. Fish fed live Artemia exposed to zinc chloride in water. Endpoints are growth and survival.
				LOAEL	2,000		--	--	--		Fish fed at same dose of zinc with 0.5% calcium experienced no adverse effects. Endpoint is growth.

Notes

AWQC = ambient water quality criteria
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
TRV = toxicity reference value
-- = Risks were not evaluated using lines of evidence requiring this information.

a - Includes AWQC and TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - Windward (2011). Values presented are lowest NOAEC with a bounded LOAEC.
c - Steevens et al. (2005)
d - See Table B-11
e - TRV is an LC_{50} divided by an uncertainty factor of 10
f - Hatakayama and Yasuo (1987), as cited in Windward (2011b)
g - Windward (2011b)
h - Windward (2011b)
i - Matta et al. (2001)
j - Hunt et al. (2002)
k - USEPA (1988) Ambient Water Quality Criteria Document for Nickel
l - Windward (2007)

Table B-16
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	2	a	Reproduction	Geometric mean of NOAELs for 5 bird species (Table B-11)
	LOAEL	3			Geometric mean of LOAELs for 4 bird species (Table B-11)
TCDD (ingested dose)	NOAEL ng/kg-d	14	b	Hen mortality and egg mortality	Ingested dose was estimated from weekly injected dose.
	LOAEL ng/kg-d	140			
TCDD (egg concentration ng/kg ww)	NOAEL	450	c	Egg mortality	Derived from multiple studies. See Appendix B
	LOAEL	2,400			
Bis(2-ethylhexyl)phthalate	NOAEL	74.9	d	Growth	Unbounded NOAEL for body weight
	LOAEL	--			
Metals					
Cadmium	NOAEL	1.47	e	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.37		Reproduction	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Chromium	NOAEL	2.66	f	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.78			Minimum bounded LOAEL for a mortality/growth/repro endpoint
Copper	NOAEL	4.05	g	Reproduction, growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	12.1			
Lead	NOAEL	1.63	h	Reproduction	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	1.94			Lowest bounded LOAEL
Mercury	NOAEL	0.078	i	Reproduction	One dose only tested. Unbounded NOAEL for first generation.
	LOAEL	0.9	j	Reproduction	Administered as methylmercury.

Table B-16
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Nickel	NOAEL	6.71	k	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	11.5		Growth	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Vanadium	NOAEL	0.344	l	Growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	0.413		Reproduction	Lowest bounded LOAEL for survival, growth, or reproduction
Zinc	NOAEL	66.1	m	Reproduction	Geometric mean of NOAELs for reproduction and growth
	LOAEL	86.6			Lowest bounded LOAEL for survival, growth, or reproduction

Notes

EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/
LOAEL = lowest observed adverse effect level
NA = not available
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Risebrough and Anderson (1975)
b - Nosek et al. (1992a)
c - Appendix B
d - O'Shea and Stafford (1980)
e - EcoSSL (USEPA 2005b)
f - EcoSSL for Cr(III) (USEPA 2008)
g - EcoSSL (USEPA 2007a)
h - EcoSSL (USEPA 2005c)
i - Heinz (1979)
j -Hill and Schaffner (1976)
k - EcoSSL (USEPA 2007b)
l - EcoSSL (USEPA 2005d)
m - USEPA (2007c)

Table B-17
Data Used to Derive Nickel TRV for Fish

Common Name	Latin Name	SMAV (µg/L)[a]	Estimated NOEC	NOEC[b]
Mummichog (adult)	*Fundulus heteroclitus*	149,900	14,990	
Atlantic silverside (larva)	*Menldia menldia*	7,960	796	
Tidewater silverside (juvenile)	*Menldia peninsulae*	38,000	3,800	
Striped bass	*Moreone saxatilis*	21,000	2,100	
Spot (juvenile)	*Lelostomus xanthurus*	70,000	7,000	
Topsmelt	*Atherinops affinis*	26,560		3,240
Geometric Mean of NOECs			3,595	

Notes

NOEC = no-observed-effect concentration

SMAV = species mean acute value

TRV = toxicity reference value

a - USEPA (1988). Ambient ALC for nickel

b - Hunt et al. (2002)

APPENDIX C
EXPOSURE POINT CONCENTRATIONS USED FOR EXPOSURE ASSESSMENT IN THE BERA

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean [b]	UCL [c]	Maximum Concentration
Sediment	Site-wide	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	103	37	94.7	160	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	103	67	0.401	0.559	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	103	90	6.69	22.9	110
		Mercury	unk	ucl.proucl.np	mg/kg	103	94	0.117	0.258	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	103	95	6.08	7.90	17.8
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	132	100	2,390	5,660	58,300
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	132	100	776	1,840	20,400
		$TEQ_{P\ B}$ [g]	unk	ucl.proucl.np	ng/kg	18	94	4.85	13.9	28.0
		$TEQ_{P\ M}$ [h]	unk	ucl.proucl.np	ng/kg	18	94	0.902	2.23	4.50
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	18	0	3,180	13,100	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	103	100	31.8	97.6	305
	Peninsula shoreline	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	31	68	158	388	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	31	58	0.342	0.710	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	31	90	6.69	35.8	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	31	94	0.270	0.707	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	31	87	5.20	8.32	14.4
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	35	100	5,470	13,300	43,900
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	35	100	1,780	4,280	12,600
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	4	100	18.4	31.6	28.0
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	4	100	2.99	4.63	4.50
		Total PCBs (sum of Aroclors) [i]	normal	ucl.t	µg/kg	4	0	14,000	35,300	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	31	100	28.0	153	228
	Shoreline	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	44	61	119	285	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	44	66	0.342	0.608	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	44	86	5.83	25.4	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	44	93	0.197	0.512	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	44	91	5.28	7.76	14.4
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	48	100	3,990	9,850	43,900
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	48	100	3,430	8,480	38,700
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	48	100	1,300	3,180	12,600
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	4	100	18.4	31.6	28.0
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	4	100	2.99	4.63	4.50
		Total PCBs (sum of Aroclors) [i]	normal	ucl.t	µg/kg	4	0	14,000	35,300	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	44	100	25.9	111	228

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment (continued)	All Outside Western Cell	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	96	32	71.7	134	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	96	65	0.359	0.498	1.40
		Copper	lognormal	ucl.cheb.log	mg/kg	96	90	6.08	19.1	110
		Mercury	unk	ucl.proucl.np	mg/kg	96	94	0.0507	0.0918	0.717
		Nickel	unk	ucl.proucl.np	mg/kg	96	95	5.83	7.65	17.8
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	122	100	1,160	3,830	58,300
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	122	100	997	3,290	49,200
		$TEQ_{P\ B}$ [g]	unk	ucl.proucl.np	ng/kg	17	94	3.55	11.1	28.0
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	17	0	1,010	4,190	12,500
		Zinc	lognormal	ucl.cheb.log	mg/kg	96	100	29.1	84.8	305
	All Background	$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	29	100	2.52	5.62	14.7
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	11	73	0.952	1.18	1.59
		$TEQ_{P\ M}$ [k,n]	normal	ucl.t	ng/kg	8	100	0.165	0.198	0.222
	Shoreline Background	$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	8	100	1.03	3.27	4.54
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	8	100	1.08	3.30	4.55
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	100	0.400	0.607	0.952
	Post-TCRA: All Site	$TEQ_{DF\ B}$-Median [e,l]	lognormal	ucl.cheb.log	ng/kg	103	100	12.9	149	482
		$TEQ_{P\ B}$-Median [g,l]	lognormal	ucl.cheb.log	ng/kg	15	93	0.878	1.48	2.03
	Post-TCRA: Shoreline	$TEQ_{DF\ M}$-Median [f,l]	unk	ucl.proucl.np	ng/kg	33	100	2.76	5.38	14.3
		$EcoTEQ_{DF\ B}$-Median [e,j,l]	lognormal	ucl.cheb.log	ng/kg	33	100	3.08	26.5	49.6
		$TEQ_{DF\ B}$-Median [e,l]	lognormal	ucl.cheb.log	ng/kg	33	100	3.50	29.2	54.9
Soils	North of I-10	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	32	63	42.7	496	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	32	94	0.362	0.753	1.73
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	8.99	43.3	121
		Mercury	unk	ucl.proucl.np	mg/kg	32	94	0.924	3.15	12.9
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	6.07	21.7	96.0
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	42	100	1,950	6,200	30,100
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	42	100	1,650	5,190	24,400
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	42	100	636	2,070	11,200
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	2	100	2.16	7.05	2.93
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	2	100	0.748	2.97	1.10
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	5	20	90.0	110	108
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	46.2	257	328

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Soils (continued)	Peninsula	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	41	71	51.0	538	2,200
		Cadmium	unk	ucl.proucl.np	mg/kg	41	95	0.376	0.705	1.73
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	11.7	58.0	121
		Mercury	unk	ucl.proucl.np	mg/kg	41	95	0.733	2.48	12.9
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	6.81	22.8	96.0
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	51	100	1,610	5,130	30,100
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	51	100	526	1,720	11,200
		$TEQ_{P\ B}$ [g]	NA	max	ng/kg	1	100	2.93	2.93	2.93
		$TEQ_{P\ M}$ [h]	NA	max	ng/kg	1	100	1.10	1.10	1.10
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	24	71	47.0	79.3	119
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	67.7	516	4,160
	South of I-10, 0 to 2 feet	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	25	88	78.9	938	2,200
		Cadmium	unk	ucl.proucl.np	mg/kg	27	100	0.336	0.591	1.28
		Chromium	lognormal	ucl.cheb.log	mg/kg	27	100	15.0	37.4	70.3
		Copper	lognormal	ucl.cheb.log	mg/kg	27	100	26.2	118	651
		Lead	lognormal	ucl.cheb.log	mg/kg	27	100	27.8	70.3	137
		Mercury	unk	ucl.proucl.np	mg/kg	27	96	0.0413	0.0765	0.156
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	30	100	6.23	31.2	105
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	30	100	5.82	16.7	38.8
		Thallium	lognormal	ucl.cheb.log	mg/kg	27	63	2.61	6.15	9.80
		Total PCBs (sum of Aroclors) [i]	lognormal	ucl.cheb.log	µg/kg	27	78	30.5	111	427
		Zinc	lognormal	ucl.cheb.log	mg/kg	27	100	178	1,260	4,160
	South of I-10, 0 to 6 inches	Barium	normal	ucl.t	mg/kg	10	100	163	226	413
		Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	10	100	92.2	586	2,200
		Cadmium	lognormal	ucl.cheb.log	mg/kg	10	100	0.262	0.946	1.28
		Chromium	lognormal	ucl.cheb.log	mg/kg	10	100	13.3	38.1	70.3
		Copper	normal	ucl.t	mg/kg	10	100	40.5	61.2	121
		Lead	lognormal	ucl.cheb.log	mg/kg	10	100	29.1	77.9	113
		Mercury	lognormal	ucl.cheb.log	mg/kg	10	100	0.0362	0.125	0.140
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	10	100	6.78	52.3	73.1
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	10	100	11.8	18.1	31.1
		Thallium	normal	ucl.t	mg/kg	10	80	5.08	6.77	9.80
		Total PCBs (sum of Aroclors) [i]	lognormal	ucl.cheb.log	µg/kg	10	80	27.6	85.7	119
		Vanadium	normal	ucl.t	mg/kg	10	100	19.7	24.5	33.9
		Zinc	lognormal	ucl.cheb.log	mg/kg	10	100	234	1420	4,160

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Soils (continued)	Post-TCRA, North of I-10	Mercury-Median [l]	unk	ucl.proucl.np	mg/kg	24	91	1.01	3.97	12.9
		Zinc-Median [l]	lognormal	ucl.cheb.log	mg/kg	24	100	43.7	253	328
		$EcoTEQ_{DF\ B}$ [e,j,l]	lognormal	ucl.cheb.log	ng/kg	34	100	4.15	20.7	33.5
		$TEQ_{DF\ B}$ [e,l]	lognormal	ucl.cheb.log	ng/kg	34	100	4.44	22.9	34.0
	Background, North of I-10	Mercury	unk	ucl.proucl.np	mg/kg	20	100	0.0422	0.0739	0.137
		Zinc	lognormal	ucl.cheb.log	mg/kg	19	100	30.6	95.7	276
		$EcoTEQ_{DF\ B}$ [e,j,l]	unk	ucl.proucl.np	ng/kg	20	100	1.75	3.88	7.75
		$TEQ_{DF\ B}$ [e,l]	unk	ucl.proucl.np	ng/kg	20	100	1.82	3.97	7.81
Surface water	Site-wide	$TEQ_{DF\ B}$ [e,p]	NA	NA	mg/L	2	100	2.63E-08	4.00E-08	4.00E-08
Common rangia	Site-wide	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	25	0	982	1,020	1,170
		Cadmium	normal	ucl.t	mg/kg	25	100	0.246	0.257	0.303
		Copper	unk	ucl.proucl.np	mg/kg	25	100	24.6	32.6	41.4
		Mercury	normal	ucl.t	mg/kg	25	92	0.0960	0.108	0.168
		Nickel	normal	ucl.t	mg/kg	25	100	12.0	13.2	20.1
		$EcoTEQ_{DF\ B}$ [e,j]	lognormal	ucl.cheb.log	ng/kg	25	100	82.2	335	928
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	25	100	90.4	369	1,020
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	25	100	9.75	43.1	108
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	25	100	23.3	93.8	254
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	25	100	27.0	43.8	64.9
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg ww	25	100	2.91	4.52	7.40
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	25	100	3.27	5.61	15.8
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	25	100	239	356	555
		Zinc	normal	ucl.t	mg/kg	25	100	95.9	99.3	119
	Peninsula only	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	15	0	979	1,050	1,170
		Cadmium	normal	ucl.t	mg/kg	15	100	0.249	0.265	0.303
		Copper	unk	ucl.proucl.np	mg/kg	15	100	29.3	39.5	41.4
		Mercury	normal	ucl.t	mg/kg	15	100	0.105	0.122	0.168
		Nickel	unk	ucl.proucl.np	mg/kg	15	100	11.5	14.9	14.9
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	15	100	123	789	1,020
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	15	100	30.7	192	254
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	15	100	29.7	55.4	64.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	15	100	3.53	4.15	5.92
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	15	100	294	463	555
		Zinc	normal	ucl.t	mg/kg	15	100	95.9	98.6	105

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia (continued)	Background	$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	10	100	1.48	1.93	2.65
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	10	100	1.19	1.41	1.93
		$EcoTEQ_{DF\ B}$ [e,j]	normal	ucl.t	ng/kg	10	100	13.6	17.6	22.5
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	10	100	14.4	18.7	23.7
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	10	100	3.51	4.50	6.65
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	10	100	11.5	13.4	17.3
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	10	100	1.74	1.98	2.52
Gulf killifish	Site-wide	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	10	0	434	439	445
		Cadmium [n]	normal	ucl.t	mg/kg	8	13	0.00794	0.00919	0.0109
		Copper	normal	ucl.t	mg/kg	10	100	5.75	6.24	7.15
		Mercury	normal	ucl.t	mg/kg	10	100	0.202	0.263	0.372
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	1.89	2.57	3.40
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	10	70	2.66	51.5	59.0
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	10	70	0.645	12.5	14.3
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	10	70	1.45	24.1	41.7
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	10	100	9.39	17.3	17.9
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	10	100	2.51	3.20	4.31
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	10	100	2.81	8.54	12.1
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	10	100	201	484	588
		Zinc	normal	ucl.t	mg/kg	10	100	174	180	195
	Peninsula only	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	6	0	432	441	445
		Cadmium [n,o]	normal	ucl.t	mg/kg	4	25	0.00755	0.0104	0.0109
		Copper	normal	ucl.t	mg/kg	6	100	6.04	6.63	7.15
		Mercury	normal	ucl.t	mg/kg	6	100	0.221	0.294	0.331
		Nickel	lognormal	ucl.cheb.log	mg/kg	6	100	2.06	3.16	3.40
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	6	100	10.4	56.9	59.0
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	6	100	4.70	38.1	41.7
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	6	100	12.9	16.9	17.9
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	6	100	4.09	13.6	12.1
		Total PCBs [m]	lognormal	ucl.cheb.log	µg/kg	6	100	199	686	588
		Zinc	normal	ucl.t	mg/kg	6	100	178	187	195

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Gulf killifish (continued)	Background	$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	8	88	0.258	0.401	0.636
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg ww	8	100	0.564	0.903	0.950
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	8	88	1.07	1.66	2.64
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	88	0.54	0.82	1.25
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	8	100	2.35	3.73	3.94
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	8	100	1.01	2.44	2.71
Blue Crab	Site-wide	Bis(2-ethylhexyl)phthalate [n]	unk	ucl.proucl.np	µg/kg	8	100	891	1,260	1,330
		Cadmium	normal	ucl.t	mg/kg	9	100	0.270	0.305	0.368
		Copper	normal	ucl.t	mg/kg	9	100	46.2	51.3	58.8
		Mercury	normal	ucl.t	mg/kg	9	100	0.0743	0.0854	0.0979
		Nickel	lognormal	ucl.cheb.log	mg/kg	9	100	1.24	2.79	3.66
		$EcoTEQ_{DF\ B}$ [e,j]	lognormal	ucl.cheb.log	ng/kg	9	100	19.5	38.9	40.8
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	9	100	22.2	44.7	46.3
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	9	100	7.80	10.6	14.9
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	9	100	17.9	22.3	26.8
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	9	100	6.52	13.8	14.5
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	9	100	5.27	6.55	8.27
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	9	100	2.33	3.00	4.00
		Total PCBs [m]	normal	ucl.t	µg/kg	9	100	69.4	80.4	100
		Zinc	normal	ucl.t	mg/kg	9	100	112	117	123
	Background[p]	$TEQ_{DF\ B}$ [e]	NA	max	ng/kg ww	3	100	0.355	0.453	0.453
		$TEQ_{P\ B}$ [g]	NA	max	ng/kg ww	3	100	0.480	0.616	0.616
		$EcoTEQ_{DF\ B}$ [e,j]	normal	max	ng/kg	3	100	1.11	1.42	1.42
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	3	100	1.18	1.55	1.55
		$TEQ_{DF\ M}$ [f]	normal	max	ng/kg	3	100	0.59	0.71	0.71
		$TEQ_{P\ B}$ [g]	normal	max	ng/kg	3	100	1.72	2.08	2.08
		$TEQ_{P\ M}$ [h]	normal	max	ng/kg	3	100	0.45	0.59	0.59

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Hardhead catfish	Site-wide	Bis(2-ethylhexyl)phthalate [n]	normal	ucl.t	µg/kg	5	80	757	1,030	1,090
		Cadmium	normal	ucl.t	mg/kg	10	80	0.0222	0.0274	0.0378
		Copper	lognormal	ucl.cheb.log	mg/kg	10	100	1.69	2.80	4.27
		Mercury	normal	ucl.t	mg/kg	10	100	0.245	0.304	0.432
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	0.937	2.46	4.36
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	10	100	79.5	93.6	117
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	10	100	27.2	32.7	44.3
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	10	100	68.7	81.6	104
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	10	100	35.4	40.3	47.4
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	10	100	12.1	14.0	17.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	10	100	22.9	27.0	30.6
		Total PCBs [m]	normal	ucl.t	µg/kg	10	100	1,480	1,680	2,010
		Zinc	normal	ucl.t	mg/kg	10	100	650	740	876
	Background	$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	8	100	2.65	3.16	3.62
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	8	100	4.84	5.44	6.56
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	8	100	7.73	9.13	10.6
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	100	6.51	7.58	8.11
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	8	100	14.2	15.8	18.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	8	100	7.80	9.54	12.4
Terrestrial invertebrates[r]	North of I-10	Cadmium	unk	ucl.proucl.np	mg/kg	32	94	3.69	6.61	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	4.63	22.3	NA
		Mercury [s]	unk	ucl.proucl.np	mg/kg	32	94	0.960	2.62	9.03
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	0.759	2.71	NA
		Total PCBs [i]	unk	ucl.proucl.np	mg/kg	5	20	0.155	0.198	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	301	528	NA
		$EcoTEQ_{DF\ B}$ [j,q]	unk	ucl.proucl.np	ng/kg	42	100	60.7	181	900
		$TEQ_{DF\ B}$ [q]	unk	ucl.proucl.np	ng/kg	42	100	117	359	1,840
	Peninsula only	Cadmium	unk	ucl.proucl.np	mg/kg	41	95	3.81	6.27	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	6.03	29.9	NA
		Mercury [r]	unk	ucl.proucl.np	mg/kg	41	95	0.788	2.10	9.03
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	0.851	2.85	NA
		Total PCBs [i]	unk	ucl.proucl.np	mg/kg	24	71	0.0639	0.130	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	341	664	NA
		$TEQ_{DF\ M}$ [q]	unk	ucl.proucl.np	ng/kg	51	100	94.2	284	1770

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Terrestrial plants[t]	North of I-10	Cadmium	unk	ucl.proucl.np	mg/kg	32	94	0.357	0.533	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	1.22	2.26	NA
		Mercury	unk	ucl.proucl.np	mg/kg	32	94	0.0347	0.118	NA
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	0.417	1.08	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	40.4	105	NA
	Peninsula	Cadmium	unk	ucl.proucl.np	mg/kg	41	95	0.365	0.514	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	1.35	2.54	NA
		Mercury	unk	ucl.proucl.np	mg/kg	41	95	0.0275	0.0930	NA
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	0.455	1.12	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	49.9	154	NA
Aquatic plants[t]	Shoreline	Cadmium	unk	ucl.proucl.np	mg/kg	44	66	0.346	0.474	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	44	86	1.03	1.83	NA
		Mercury	unk	ucl.proucl.np	mg/kg	44	93	0.00740	0.0192	NA
		Nickel	unk	ucl.proucl.np	mg/kg	44	91	0.376	0.502	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	44	100	29.3	65.7	NA

Notes

BERA - Baseline Ecological Risk Assessment
CT = central tendency
EPC = exposure point concentration
NA = not applicable
RM = reasonable maximum
ROS = regression on order statistics, a method for substituting for non-detects
TCRA = time critical removal action
UCL = upper confidence limit on the mean
ucl.t = UCL for normally distributed data, calculated based on the T statistic
ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor
ucl.proucl.np = nonparametric UCL for an unknown data distribution, method is based on that used in ProUCL (USEPA 2010)
unk = unknown distribution

a - All concentrations are on a dry weight basis unless the units indicate otherwise.
b - The mean value is the CT EPC.
c - The UCL will be used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration will be selected as the RM EPC.
d - Because the detection frequency was between 20 and 50% and N > 10, ROS was used for calculating the UCL.
e - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.
f - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
g - Toxicity equivalent for polychlorinated biphenyls calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.
h - Toxicity equivalent for polychlorinated biphenyls calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
i - Sum of total Aroclors with nondetects set at one-half the detection limit.
j - Calculated using a relative bioavailability adjustment factor for avian receptors for 2,3,7,8-TCDD.
k - Toxicity equivalent for polychlorinated biphenyls calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
l - Median of background data (soils or sediments, as appropriate) used to substitute for samples in the TCRA footprint.
m - Sum of 43 PCB congeners with non-detects set at one-half the detection limit.
n - High-biasing nondetects (nondetects > highest detected value) were removed prior to calculating the EPC.
o - Detection frequency between 20 and 50% but N<10, so no ROS performed for this dataset.
p - Data set too small to generate a UCL; maximum value is used for the RM.
q - Estimated using site soil data and regression relationships (see Appendix D).
r - Unless otherwise footnoted, estimated using site data for soils from indicated exposure area and soil-to-invertebrate BAFs
s - Estimated using site soil data and variable BAFs depending on concentration in soils (Burton et al. 2006)
t - Estimated using site data for soils or sediments from indicated exposure area and soil-to-plant BAFs

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment	FCA1	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	29	52	33.2	55.8	120
		Cadmium	lognormal	ucl.cheb.log	mg/kg	29	79	0.345	0.815	1.40
		Copper	lognormal	ucl.cheb.log	mg/kg	29	90	7.51	31.9	110
		Mercury	lognormal	ucl.cheb.log	mg/kg	29	93	0.0277	0.0992	0.0960
		Nickel	lognormal	ucl.cheb.log	mg/kg	29	97	5.16	12.7	15.7
		Zinc	lognormal	ucl.cheb.log	mg/kg	29	100	39.5	132	305
	FCA2	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	57	37	103	162	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	57	63	0.374	0.602	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	57	93	7.18	21.4	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	57	95	0.179	0.428	2.02
		Nickel	lognormal	ucl.cheb.log	mg/kg	57	100	4.61	9.64	17.8
		Zinc	lognormal	ucl.cheb.log	mg/kg	57	100	32.1	88.5	228
	FCA3	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	17	12	192	957	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	17	59	0.394	0.725	1.00
		Copper	lognormal	ucl.cheb.log	mg/kg	17	82	4.34	33.3	19.0
		Mercury	unk	ucl.proucl.np	mg/kg	17	94	0.0420	0.0792	0.0900
		Nickel	unk	ucl.proucl.np	mg/kg	17	76	5.33	10.1	11.5
		Zinc	lognormal	ucl.cheb.log	mg/kg	17	100	21.3	167	97.2
Common rangia	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	1,050	1,110	1,130
		Cadmium	normal	ucl.t	mg/kg	5	100	0.257	0.283	0.290
		Copper	normal	ucl.t	mg/kg	5	100	17.4	18.3	18.8
		Mercury	normal	ucl.t	mg/kg	5	100	0.0946	0.117	0.127
		Nickel	normal	ucl.t	mg/kg	5	100	16.5	19.1	20.1
		Zinc	normal	ucl.t	mg/kg	5	100	107	116	119
	FCA2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	15	0	913	960	1,100
		Cadmium	normal	ucl.t	mg/kg	15	100	0.237	0.252	0.303
		Copper	lognormal	ucl.cheb.log	mg/kg	15	100	22.8	35.6	41.4
		Mercury	normal	ucl.t	mg/kg	15	87	0.0837	0.100	0.136
		Nickel	normal	ucl.t	mg/kg	15	100	10.4	11.7	14.2
		Zinc	normal	ucl.t	mg/kg	15	100	93.1	97.2	105

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia (continued)	FCA3	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	5	0	1,120	1,260	1,170
		Cadmium	normal	ucl.t	mg/kg	5	100	0.262	0.286	0.290
		Copper	normal	ucl.t	mg/kg	5	100	32.0	32.5	32.7
		Mercury	normal	ucl.t	mg/kg	5	100	0.134	0.154	0.168
		Nickel	normal	ucl.t	mg/kg	5	100	12.3	14.9	14.9
		Zinc	normal	ucl.t	mg/kg	5	100	92.8	98.6	101
Gulf killifish	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	443
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00940	N/A	0.00970
		Copper	normal	ucl.t	mg/kg	2	100	5.43	N/A	5.58
		Mercury	normal	ucl.t	mg/kg	2	100	0.117	N/A	0.136
		Nickel	normal	ucl.t	mg/kg	2	100	1.58	N/A	1.63
		Zinc	normal	ucl.t	mg/kg	2	100	168	N/A	176
	FCA2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	6	0	431	440	445
		Cadmium [e,f]	normal	ucl.t	mg/kg	4	25	0.00789	0.0104	0.0109
		Copper	normal	ucl.t	mg/kg	6	100	5.71	6.60	7.15
		Mercury	normal	ucl.t	mg/kg	6	100	0.206	0.302	0.372
		Nickel	normal	ucl.t	mg/kg	6	100	1.80	1.94	2.03
		Zinc	normal	ucl.t	mg/kg	6	100	171	174	176
	FCA3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	439
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00657	N/A	0.00771
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	6.40
		Mercury	normal	ucl.t	mg/kg	2	100	0.278	N/A	0.319
		Nickel	normal	ucl.t	mg/kg	2	100	2.73	N/A	3.40
		Zinc	normal	ucl.t	mg/kg	2	100	191	N/A	195
Blue crab	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	3	100	814	N/A	894
		Cadmium	normal	ucl.t	mg/kg	3	100	0.273	N/A	0.300
		Copper	normal	ucl.t	mg/kg	3	100	45.3	N/A	55.8
		Mercury	normal	ucl.t	mg/kg	3	100	0.0888	N/A	0.0979
		Nickel	normal	ucl.t	mg/kg	3	100	2.51	N/A	3.66
		Zinc	normal	ucl.t	mg/kg	3	100	117	N/A	122

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Blue crab (continued)	FCA2	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	2	100	970	N/A	1,170
		Cadmium	normal	ucl.t	mg/kg	3	100	0.318	N/A	0.368
		Copper	normal	ucl.t	mg/kg	3	100	48.1	N/A	58.8
		Mercury	normal	ucl.t	mg/kg	3	100	0.0568	N/A	0.0693
		Nickel	normal	ucl.t	mg/kg	3	100	0.829	N/A	1.03
		Zinc	normal	ucl.t	mg/kg	3	100	113	N/A	123
	FCA3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	3	100	914	N/A	1,330
		Cadmium	normal	ucl.t	mg/kg	3	100	0.220	N/A	0.274
		Copper	normal	ucl.t	mg/kg	3	100	45.3	N/A	48.9
		Mercury	normal	ucl.t	mg/kg	3	100	0.0772	N/A	0.0974
		Nickel	normal	ucl.t	mg/kg	3	100	1.13	N/A	1.74
		Zinc	normal	ucl.t	mg/kg	3	100	106	N/A	112
Hardhead catfish	FCA1	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	2	100	999	N/A	1,090
		Cadmium	normal	ucl.t	mg/kg	3	100	0.0264	N/A	0.0378
		Copper	normal	ucl.t	mg/kg	3	100	1.72	N/A	2.31
		Mercury	normal	ucl.t	mg/kg	3	100	0.300	N/A	0.338
		Nickel	normal	ucl.t	mg/kg	3	100	0.786	N/A	1.22
		Zinc	normal	ucl.t	mg/kg	3	100	762	N/A	876
	FCA2	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	3	67	595	N/A	833
		Cadmium	normal	ucl.t	mg/kg	4	75	0.0206	0.0300	0.0289
		Copper	normal	ucl.t	mg/kg	4	100	1.39	1.72	1.78
		Mercury	normal	ucl.t	mg/kg	4	100	0.252	0.405	0.432
		Nickel	normal	ucl.t	mg/kg	4	100	0.846	1.43	1.49
		Zinc	normal	ucl.t	mg/kg	4	100	528	711	748

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Hardhead catfish (continued)	FCA3	Bis(2-ethylhexyl)phthalate [e,g]	normal	ucl.t	µg/kg	3	0	331	N/A	331
		Cadmium	normal	ucl.t	mg/kg	3	67	0.0202	N/A	0.0323
		Copper	normal	ucl.t	mg/kg	3	100	2.53	N/A	4.27
		Mercury	normal	ucl.t	mg/kg	3	100	0.181	N/A	0.254
		Nickel	normal	ucl.t	mg/kg	3	100	2.21	N/A	4.36
		Zinc	normal	ucl.t	mg/kg	3	100	701	N/A	782

Notes

CT = central tendency
EPC = exposure point concentration
FCA = fish collection area
N/A = not applicable
RM = reasonable maximum
ROS = regression on order statistics, a method for substituting for non-detects
UCL = upper confidence limit on the mean
ucl.t = UCL for normally distributed data, calculated based on the T statistic
ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor
ucl.proucl.np = nonparametric UCL for an unknown data distribution, same method as ProUCL (USEPA 2010)
unk = unknown distribution
a - All concentrations are on a dry weight basis unless the units indicate otherwise.
b - The mean value is the CT EPC.
c - The UCL will be used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration will be selected as the RM EPC. For N ≤ 3, a UCL cannot be calculated, so the average of the two samples and the maximum value only are reported in these cases.
d - Because the detection frequency was between 20 and 50% and N >10, ROS was used for calculating the UCL.
e - High-biasing nondetects (nondetects > highest detected value) were removed prior to calculating the EPC.
f - Detection frequency between 20 and 50% but N<10, so no ROS performed for this data set.
g - The highest detection limit for all FCAs was used to generate statistics.

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment	TTR1 and TTR2 [d]	Bis(2-ethylhexyl)phthalate [e]	lognormal	ucl.cheb.log	µg/kg	7	43	22.5	84.4	73.0
		Cadmium	normal	ucl.t	mg/kg	7	100	0.431	0.542	0.700
		Copper	normal	ucl.t	mg/kg	7	100	7.82	10.1	11.5
		Mercury	normal	ucl.t	mg/kg	7	100	0.0330	0.0445	0.0560
		Nickel	normal	ucl.t	mg/kg	7	100	7.60	9.49	12.0
		Zinc	normal	ucl.t	mg/kg	7	100	39.4	54.4	65.0
	TTR3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	10	90	180	249	350
		Cadmium	unk	ucl.proucl.np	mg/kg	10	60	0.346	1.01	1.50
		Copper	lognormal	ucl.cheb.log	mg/kg	10	100	10.6	30.1	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	10	100	0.295	1.16	2.02
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	4.26	9.42	14.1
		Zinc	lognormal	ucl.cheb.log	mg/kg	10	100	38.2	113	197
	TTR4	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	6	0	11.4	18.9	20.0
		Cadmium [e]	unk	ucl.proucl.np	mg/kg	6	33	0.200	0.556	0.600
		Copper	lognormal	ucl.cheb.log	mg/kg	6	50	1.73	8.94	15.2
		Mercury	unk	ucl.proucl.np	mg/kg	6	100	0.0153	0.0493	0.0530
		Nickel	lognormal	ucl.cheb.log	mg/kg	6	100	1.80	6.48	10.0
		Zinc	lognormal	ucl.cheb.log	mg/kg	6	100	11.6	51.6	74.4
	TTR5	Bis(2-ethylhexyl)phthalate [f]	lognormal	ucl.cheb.log	µg/kg	12	33	19	37.1	76.0
		Cadmium	unk	ucl.proucl.np	mg/kg	12	50	0.138	0.383	0.700
		Copper	normal	ucl.t	mg/kg	12	100	6.08	7.53	11.8
		Mercury	lognormal	ucl.cheb.log	mg/kg	12	83	0.0130	0.0330	0.0520
		Nickel	lognormal	ucl.cheb.log	mg/kg	12	100	5.18	8.61	12.5
		Zinc	lognormal	ucl.cheb.log	mg/kg	12	100	20.9	48.3	55.4
	TTR6	Bis(2-ethylhexyl)phthalate	all below DL	max	µg/kg	5	0	9.50	9.50	9.50
		Cadmium	all below DL	max	mg/kg	5	0	0.100	0.100	0.100
		Copper [e]	lognormal	ucl.cheb.log	mg/kg	5	40	0.812	4.26	3.50
		Mercury	normal	ucl.t	mg/kg	5	100	0.00590	0.0104	0.0140
		Nickel	normal	ucl.t	mg/kg	5	20	0.315	0.377	0.425
		Zinc	lognormal	ucl.cheb.log	mg/kg	5	100	3.35	8.61	9.00

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia	TTR1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	1,050	1,110	1,130
		Cadmium	normal	ucl.t	mg/kg	5	100	0.257	0.283	0.290
		Copper	normal	ucl.t	mg/kg	5	100	17.4	18.3	18.8
		Mercury	normal	ucl.t	mg/kg	5	100	0.0946	0.117	0.127
		Nickel	normal	ucl.t	mg/kg	5	100	16.5	19.1	20.1
		Zinc	normal	ucl.t	mg/kg	5	100	107	116	119
	TTR3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	954	1,050	1,100
		Cadmium	lognormal	ucl.cheb.log	mg/kg	5	100	0.274	0.321	0.303
		Copper	normal	ucl.t	mg/kg	5	100	38.1	41.2	41.4
		Mercury	normal	ucl.t	mg/kg	5	100	0.124	0.135	0.136
		Nickel	unk	ucl.proucl.np	mg/kg	5	100	12.5	16.8	14.2
		Zinc	normal	ucl.t	mg/kg	5	100	98.8	104	105
	TTR4	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	920	985	995
		Cadmium	normal	ucl.t	mg/kg	5	100	0.226	0.243	0.252
		Copper	normal	ucl.t	mg/kg	5	100	17.6	19.8	19.8
		Mercury	normal	ucl.t	mg/kg	5	60	0.0700	0.101	0.108
		Nickel	lognormal	ucl.cheb.log	mg/kg	5	100	8.90	11.8	11.7
		Zinc	normal	ucl.t	mg/kg	5	100	84.6	92.5	97.6
	TTR5	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	865	995	1,000
		Cadmium	normal	ucl.t	mg/kg	5	100	0.210	0.227	0.227
		Copper	lognormal	ucl.cheb.log	mg/kg	5	100	17.7	21.3	21.1
		Mercury	normal	ucl.t	mg/kg	5	100	0.0568	0.0666	0.0657
		Nickel	normal	ucl.t	mg/kg	5	100	9.57	13.0	13.3
	TTR6	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	5	0	1120	1260	1170
		Cadmium	normal	ucl.t	mg/kg	5	100	0.262	0.286	0.290
		Copper	normal	ucl.t	mg/kg	5	100	32.0	32.5	32.7
		Mercury	normal	ucl.t	mg/kg	5	100	0.134	0.154	0.168
		Nickel	normal	ucl.t	mg/kg	5	100	12.3	14.9	14.9
		Zinc	normal	ucl.t	mg/kg	5	100	92.8	98.6	101

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Gulf killifish	TTR2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	443
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00940	N/A	0.00970
		Copper	normal	ucl.t	mg/kg	2	100	5.43	N/A	5.58
		Mercury	normal	ucl.t	mg/kg	2	100	0.117	N/A	0.136
		Nickel	normal	ucl.t	mg/kg	2	100	1.58	N/A	1.63
		Zinc	normal	ucl.t	mg/kg	2	100	168	N/A	176
	TTR3[g]	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	439	N/A	445
		Copper	normal	ucl.t	mg/kg	2	100	5.75	N/A	6.16
		Mercury	normal	ucl.t	mg/kg	2	100	0.251	N/A	0.331
		Nickel	normal	ucl.t	mg/kg	2	100	1.99	N/A	2.03
		Zinc	normal	ucl.t	mg/kg	2	100	170	N/A	175
	TTR4	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	435	N/A	436
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00724	N/A	0.00806
		Copper	normal	ucl.t	mg/kg	2	100	5.20	N/A	6.36
		Mercury	normal	ucl.t	mg/kg	2	100	0.232	N/A	0.372
		Nickel	normal	ucl.t	mg/kg	2	100	1.73	N/A	1.74
		Zinc	normal	ucl.t	mg/kg	2	100	170	N/A	171
	TTR5	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	418	N/A	420
		Cadmium	normal	ucl.t	mg/kg	2	50	0.00853	N/A	0.0109
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	7.15
		Mercury	normal	ucl.t	mg/kg	2	100	0.135	N/A	0.145
		Nickel	normal	ucl.t	mg/kg	2	100	1.68	N/A	1.81
		Zinc	normal	ucl.t	mg/kg	2	100	171	N/A	176

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
	TTR6	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	439
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00657	N/A	0.00771
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	6.40
		Mercury	normal	ucl.t	mg/kg	2	100	0.278	N/A	0.319
		Nickel	normal	ucl.t	mg/kg	2	100	2.73	N/A	3.40
		Zinc	normal	ucl.t	mg/kg	2	100	191	N/A	195

Notes

CT = central tendency

EPC = exposure point concentration

N/A = not applicable

RM = reasonable maximum

ROS = regression on order statistics, a method for substituting for non-detects

TTR = transect

UCL = upper confidence limit on the mean

ucl.t = UCL for normally distributed data, calculated based on the T statistic

ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor

ucl.proucl.np = nonparametric UCL for an unknown data distribution, same method as ProUCL (USEPA 2010)

unk= unknown distribution

a - All concentrations are on a dry weight basis unless the units indicate otherwise.

b - The mean value will be used as the CT EPC.

c - The UCL is used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration is selected as the RM EPC. For N ≤ 3, a UCL cannot be calculated, so the average of the two samples and the maximum value only are reported in these cases.

d - Transects 1 and 2 had too much overlap to create distinct data sets.

e - Detection frequency between 20 and 50% but N<10, so no ROS performed for this data set

f - Because the detection frequency was between 20 and 50% and N >10, ROS was used for calculating the UCL.

g - No cadmium data are available for Gulf killifish in transect 3 due to removal of high-biasing nondetects. Cadmium killifish data from TTR5 were used in cadmium exposure assessment for TTR3.

APPENDIX D
ESTIMATION OF DIOXIN AND FURAN CONCENTRATIONS IN TERRESTRIAL INVERTEBRATE TISSUE FOR THE EXPOSURE MODEL

ESTIMATION OF DIOXIN AND FURAN CONCENTRATIONS IN TERRESTRIAL INVERTEBRATE TISSUE FOR THE EXPOSURE MODEL

SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
$COPC_E$	chemical of potential ecological concern
CV	coefficient of variation
EPC	exposure point concentration
Site	San Jacinto River Waste Pits Superfund site
TEF	toxicity equvalency factor
TEQ	toxicity equivalent
TEQ_{DF}	toxicity equivalent for dioxins and furans
$TEQ_{DF,B}$	toxicity equivalent for dioxins and furans using avian TEFs
$TEQ_{DF,M}$	toxicity equivalent for dioxins and furans using mammalian TEFs
USEPA	U.S. Environmental Protection Agency

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1 INTRODUCTION

This Appendix describes the methods used to generate toxicity equivalent (TEQ) concentrations for dioxins and furans (TEQ_{DF}) in terrestrial invertebrate tissues for use in the baseline ecological risk assessment (BERA) exposure model. No empirical data are available for dioxins and furans in terrestrial invertebrate tissue at the Site, so a modeled approach is needed.

Unlike the approach used for other chemicals of potential ecological concern ($COPC_{ES}$) at the Site, a single regression equation cannot be used to estimate TEQ_{DF} because the individual dioxin congeners cannot be assumed to have the same rates or patterns of bioaccumulation (Matscheko 2002; MACTEC 2004; Integral 2010). A review of the scientific literature and a search of ecological risk assessment documents was conducted to identify sources of colocated soil and earthworm tissue dioxin and furan data that could be used for developing congener-specific uptake relationships. Although there are some published uptake factors (derived as the ratio of concentrations in earthworms to those in soil as for a BAF) for a few common dioxin congeners in the literature, notably 2,3,7,8-TCDD (Sample et al. 1998), available uptake factors cannot be extrapolated across all dioxin and furan congeners to estimate invertebrate tissue concentrations for all of the individual congeners. This was demonstrated in the Tittabawassee risk assessment when use of simplified uptake factors from the literature of 5 for 2,3,7,8-TCDD and 0.1 for 2,3,7,8-TCDF led to a 10-fold overprediction of dioxin and furan concentrations in soil invertebrates relative to measured values (Galbraith and MDEQ 2004, Kay et al. 2005). Other CERCLA sites have used data sets of similar sizes to evaluate soil-earthworm relationships and have shown that uptake factors are variable across congeners. For example, the Centredale ecological risk assessment (MACTEC 2004) used a data set of N=11 with detection frequencies between about 30% and 100% for individual congeners to describe a range from < 0.01 to 0.7 for soil-to-earthworm uptake factors for individual dioxin congeners (MACTEC 2004). Unfortunately, earthworms were not depurated in this study and therefore the data could not be used in this BERA to establish an estimate for uptake into tissues.

Data from a published U.S. Environmental Protection Agency-accepted study of dioxins in earthworm tissue and colocated soils from the Cass Lake Superfund site (Integral 2007) were

used to generate significant regression relationships for individual dioxin and furan congeners. These regressions were applied to this BERA using Site soils data to estimate concentrations of congeners in terrestrial invertebrate tissue. Toxicity equivalency factors (TEFs) (Van den Berg et al. 2006) were then used to calculate TEQs for individual dioxin and furan congeners that were summed to obtain TEQ_{DF}s for terrestrial invertebrates at the Site. Exposure point concentrations (EPCs) for the resulting TEQ_{DF}s were calculated, and used in the wildlife exposure models for surrogate receptors at the Site that eat terrestrial invertebrates (killdeer and raccoon).

The remainder of this Appendix describes the details of the methods used to develop statistical relationships used to estimate the TEQ_{DF} in terrestrial invertebrate tissue, and the derivation of the tissue estimates. This includes a) the derivation of relationships between soil and earthworm tissue for individual congeners; b) the estimation of dioxin and furan concentrations in tissue for those congeners without a statistically significant relationship to soil concentration; and c) the application of these relationships to calculate a TEQ_{DF} for relevant taxa and exposure areas used in the exposure model (Section 3).

2 DERIVATION OF EQUATIONS DESCRIBING ESTIMATES OF INDIVIDUAL CONGENER CONCENTRATIONS IN TERRESTRIAL INVERTEBRATES

This section describes 1) the data used to develop soil-invertebrate tissue relationships; 2) the methods used to derive regression equations for individual congeners; and 3) the methods used to select an alternative approach for estimating a congener's concentration in tissue when no statistically significant soil-to-tissue regression equation could be identified.

2.1 Data Used for Developing Soil-Invertebrate Tissue Relationships

Available literature and Superfund-related reports evaluating dioxin and furan uptake into terrestrial invertebrates, either did not provide congener-specific data or did not use methods that would allow for development of tissue-specific uptake estimates (Section 1). The data used to develop soil-invertebrate tissue relationships for this risk assessment are from a study evaluating dioxins and furans in earthworms and soils at the St. Regis Paper Company Superfund Site in Cass Lake, Minnesota (Cass Lake site) (Integral 2007). The data set

consisted of four co-located soil and earthworm tissue samples[1] and two additional soil-earthworm paired samples from a 28-day laboratory bioacummulation study. All soil samples were collected from the top 12 inches below ground surface. Undecomposed plant materials were removed from the surface prior to collection of soil samples, and a homogenized composite of five subsamples from a single location was prepared for each sampling location. Earthworm samples collected from each soil sampling location were composited into a sample of at least 50 g and were depurated in the laboratory for 24 hours to eliminate soils and other gut contents from the worms prior to analysis. Two additional soil locations were selected for the laboratory bioaccumulation study because these soils appeared suitable for invertebrates but insufficient earthworm sample mass was available at these two stations. Therefore, a 28-day earthworm bioaccumulation study using *Eisenia fetida* was used (ASTM Method E 1676-84) and dioxins and furans were measured in soils and the tested earthworm samples from these locations. These two sets of data (Table D-1) yielded six co-located samples that were used to develop regressions. All data were validated and reported for this study according to standard protocols for Superfund sites (Integral 2007).

Soil data were reported on a mg/kg dry weight basis; because the exposure model requires concentrations in receptor prey in dry weight units, earthworm data originally reported on a mg/kg wet weight basis were converted to dry weight using the following equation:

$$C_{E,dw} = C_{E,ww} \div (f_{solids})$$

Where:

$C_{E,dw}$ = concentration in the tissue of the earthworm, dry weight (mg/kg)

$C_{E,ww}$ = concentration in the tissue of the earthworm, wet weight (mg/kg)

f_{solids} = fraction of the organism that is solid material (not water).

Solids data were available for each of the samples except ECO-07. For the ECO-07 sample, the overall average of f_{solids} of the other samples was used as an estimate for this sample.

The range of most dioxin and furan concentrations in soil at the Cass Lake site were similar to the range of concentrations in soils from the San Jacinto River Waste Pits site

[1] Five colocated soil and earthworm samples (field-collected) were available from the study. However, earthworms in one sample were not depurated and had substantively higher concentrations of all congeners than other earthworms, likely attributable to soil remaining in the gut contents. This sample was therefore not included in this analysis.

(Figure D-1). In particular, dioxin and furan congener distributions in soils of Cass Lake are similar to or higher than the ranges of concentrations of congeners in those San Jacinto site soils collected from locations outside of the 1966 perimeter of the waste impoundments north of I-10 (Figure D-2). Therefore, for the majority of congeners, predictions made on the basis of Cass Lake soil concentration data are not outside of the range of San Jacinto site soils,supporting the premise that the Cass Lake dataset is appropriate for use in generating regression relationships that can be applied to the Site data. Two exceptions are 2,3,7,8-TCDD and 2,3,7,8-TCDF, which have higher concentrations in SJRWP soils within the impoundments relative to Cass Lake soil concentrations (Figure D-3). Different approaches to treatment of these congeners in modeling from soils inside and outside of the 1966 impoundment perimeter are discussed further below.

2.2 Derivation of Regressions for Estimating Concentrations of Dioxin and Furan Congeners in Invertebrate Tissue

Concentrations of individual dioxin and furan congeners in earthworm tissue from the Cass Lake datasets were regressed against soil concentration data. The distribution of the dataset was evaluated for each congener in soil and earthworm tissue, and nearly all soil and earthworm congener datasets were found to have a lognormal distribution. Both untransformed and log-transformed regressions were evaluated and in all cases, log–log relationships had similar or lower p values and similar R^2 values, supporting the assumption that log-transformed relationships are the best models for these datasets. P-values ≤ 0.1 were considered statistically significant; this p value is used because of the small size of the sample set and consequently lower power (Sokal and Rohlf 1981; Royall 1986). Significant regression relationships between soil and tissue could be developed for 11 of the 17 congeners (Table D-2). The approach taken to estimate tissue concentration for the remaining six congeners is described in the next section.

2,3,7,8-TCDD was not detected in five of six Cass Lake earthworm tissue samples. In Cass Lake soil, 2,3,7,8-TCDD was not detected in three samples. There is consequently some uncertainty regarding the soiltissue relationship for TCDD given the censored data; however, a significant regression relationship was derived for this congener (Figure D-4) when half of the detection limits were used for the undetected values. The fact that TCDD in soil was

detected with colocated tissue samples in which it was not detected suggests that the uptake rate from soil to earthworm tissues is low for this congener. Because of the uncertainties of the censored data for TCDD, this regression for TCDD was not used in evaluation of correlates for congeners without regression relationships (Section 2.3 below).

2.3 Development of Estimates Using Correlated Congeners for Cases when Significant Regressions Could Not Be Identified

For six congeners, no statistically significant relationships between soil and earthworm concentrations were identified (Table D-2). Therefore, an alternative approach was used to estimate concentrations of these congeners in earthworm tissue. Spearman correlations[2] were used to evaluate relationships of each of these congeners in earthworm tissue with each of the other 11 congeners in earthworm tissue. Those congeners that had the highest (Table D-3), statistically significant (Table D-4) correlation coefficients with each of the six congeners were further evaluated in order to select the congener that provided the best correlate for each of the six congeners of interest.

To identify the best correlate, ratios were first calculated for the colocated pairs of earthworm tissue data for the congener of interest and each of its potentially well-correlated congeners. The average ratio and the coefficient of variation (CV) was then calculated for each congener pair (Table D-5). Because more than one congener was identified as a potential correlate in all cases, the congener pair with the lowest CV was selected as the best fit (Table D-5). The concentration of the congener of interest was then estimated by taking the concentration of its selected correlate and multiplying it by the mean ratio.

As discussed above, the range of 2,3,7,8-TCDF concentrations in Cass Lake soil is similar to the range of concentrations in San Jacinto site soils outside of the waste impoundments north of I-10, but the highest Cass Lake concentrations of TCDF are lower than those from within the northern impoundments. 2,3,7,8-TCDF in Cass Lake soils was found to be significantly and strongly correlated with 1,2,3,6,7,8-HxCDD in Cass Lake tissues (Table D-3). In addition, concentrations of 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD in soils outside the impoundments are similar. This similarity indicates that use of 1,2,3,6,7,8-HxCDD as a correlate would not

[2] Spearman's non-parametric ranked correlations (rho), for evaluating statistical dependence between variables.

lead to underpredictions of 2,3,7,8-TCDF in tissue (Figure D-2). However, concentrations of 1,2,3,6,7,8-HxCDD were low relative to 2,3,7,8-TCDF in soils inside the waste impoundments (Figure D-3), so a different congener was needed for prediction of TCDF concentrations in tissue from concentrations in soils inside the impoundments. 1,2,3,4,7,8-HxCDF, also significantly correlated with 2,3,7,8-TCDF (Table D-3), has concentrations that were more consistent with 2,3,7,8-TCDF within the waste impoundment soils (Figure D-3). Therefore, this congener was applied as the selected correlate for 2,3,7,8-TCDF for soil samples within the waste impoundments.

3 ESTIMATION OF TEQS IN TERRESTRIAL INVERTEBRATE TISSUE AT THE SITE

Surface soil data for the San Jacinto Site were used with the regressions and correlated congener ratios described in Section 2 to estimate $TEQ_{DF,B}$ and $TEQ_{DF,M}$ in terrestrial invertebrates at the Site. This section describes 1) how the Site soil data was selected, and 2) how the TEQ calculations were performed to generate these estimates.

3.1 Generation of Site Soil Data for Use in TEQ Calculations

Calculation of an estimated TEQ concentration in terrestrial invertebrates requires that Site soil data be used as the input variable to the individual congener regressions. Site soil datasets selected for calculation of TEQs were surface soil samples within the exposure units identified in the BERA for upland receptors whose diets include terrestrial invertebrate prey: raccoon (Figure 4-10 in the BERA) and killdeer (Figure 4-9 in the BERA).

3.2 Calculation of TEQs and Terrestrial Invertebrate EPCs

The congener-specific regression equations (Section 2.2) and correlations (Section 2.3) were used with corresponding congener concentrations in individual soil samples (Section 3.1) to estimate individual congener concentrations in the tissue of soil invertebrates.[3] For each soil sample, the result is an estimate of the concentration of each congener in a hypothetical corresponding invertebrate sample. Resulting congener-specific concentrations in modeled earthworm tissue were then multiplied by the avian or mammalian TEF, as appropriate, to

[3] For 2,3,7,8-TCDD, an additional step was taken to adjust for known limitations in bioavailability of this congener to avian receptors, using a relative bioavailability adjustment factor. Results are presented both with and without a bioavailability adjustment factor in the uncertainty analysis. See the main text, Section 4, for details.

compute the TEQ_{DF} concentration for a modeled individual earthworm sample. The final result was a set of estimated $TEQ_{DF,B}$ and $TEQ_{DF,M}$ concentrations for modeled earthworm samples, each corresponding to a specific soil sample. From the sets of estimated earthworm $TEQ_{DF,B}$ and $TEQ_{DF,M}$ concentrations, central tendency and reasonable maximum exposure point concentrations were calculated using the approach described in Section 3.8.2 of the BERA to generate estimates of terrestrial invetebrate tissue concentrations that were required for modeling (Table D-6).

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TABLES

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-07[b]	2,3,7,8-TCDD	3.63E-07	U	9.86E-07	U
	1,2,3,7,8,9-HxCDD	6.41E-05		2.15E-05	
	1,2,3,4,7,8,9-HpCDF	3.58E-04		7.85E-06	J
	1,2,3,6,7,8-HxCDD	5.39E-04		4.87E-05	
	1,2,3,7,8-PCDD	1.12E-05		4.77E-06	JEMPC
	1,2,3,4,6,7,8-HpCDD	2.65E-02		6.63E-04	
	OCDD	4.74E-01	J	4.09E-03	
	2,3,7,8-TCDF	1.13E-05		8.49E-07	U
	1,2,3,7,8,9-HxCDF	7.34E-05		3.90E-06	U
	2,3,4,7,8-PCDF	7.72E-05		6.01E-06	J
	1,2,3,4,7,8-HxCDD	2.67E-07	U	1.03E-05	JEMPC
	1,2,3,6,7,8-HxCDF	7.10E-05		8.81E-06	J
	1,2,3,7,8-PCDF	5.25E-05	EMPC	1.15E-05	
	2,3,4,6,7,8-HxCDF	1.47E-04		9.83E-06	J
	1,2,3,4,6,7,8-HpCDF	3.19E-03		2.11E-04	
	1,2,3,4,7,8-HxCDF	3.67E-04		2.47E-05	
	OCDF	3.77E-02		3.54E-04	
ECO-08	2,3,7,8-TCDD	2.41E-07	U	6.45E-07	JEMPC
	1,2,3,7,8,9-HxCDD	1.24E-04	EMPC	8.75E-05	
	1,2,3,4,7,8,9-HpCDF	4.40E-04		2.53E-04	
	1,2,3,6,7,8-HxCDD	4.25E-04	EMPC	4.17E-04	
	1,2,3,7,8-PCDD	1.61E-05		2.16E-05	
	1,2,3,4,6,7,8-HpCDD	1.41E-02		6.18E-03	
	OCDD	2.07E-01	J	5.62E-02	J
	2,3,7,8-TCDF	4.10E-06		3.32E-06	
	1,2,3,7,8,9-HxCDF	2.97E-06	U	4.20E-06	U
	2,3,4,7,8-PCDF	4.88E-05		2.64E-05	
	1,2,3,4,7,8-HxCDD	2.63E-05	EMPC	5.15E-05	
	1,2,3,6,7,8-HxCDF	1.47E-04		8.49E-05	
	1,2,3,7,8-PCDF	6.20E-05		3.86E-05	
	2,3,4,6,7,8-HxCDF	2.19E-04		1.10E-04	
	1,2,3,4,6,7,8-HpCDF	3.92E-03		2.13E-03	
	1,2,3,4,7,8-HxCDF	3.95E-04		2.40E-04	
	OCDF	1.89E-02		5.06E-03	

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-09	2,3,7,8-TCDD	1.98E-07	U	1.38E-07	U
	1,2,3,7,8,9-HxCDD	1.62E-06	JEMPC	1.19E-05	
	1,2,3,4,7,8,9-HpCDF	6.95E-07	U	1.15E-05	
	1,2,3,6,7,8-HxCDD	4.30E-06		2.27E-05	
	1,2,3,7,8-PCDD	1.88E-07	U	2.21E-06	J
	1,2,3,4,6,7,8-HpCDD	1.79E-04		7.24E-04	
	OCDD	1.77E-03	J	6.67E-03	J
	2,3,7,8-TCDF	1.64E-07	U	1.12E-06	U
	1,2,3,7,8,9-HxCDF	2.05E-07	U	9.78E-08	U
	2,3,4,7,8-PCDF	1.77E-07	U	1.98E-06	J
	1,2,3,4,7,8-HxCDD	3.26E-07	U	5.27E-06	
	1,2,3,6,7,8-HxCDF	9.22E-07	JEMPC	5.02E-06	
	1,2,3,7,8-PCDF	1.82E-07	U	2.33E-06	J
	2,3,4,6,7,8-HxCDF	1.46E-06	J	6.94E-06	
	1,2,3,4,6,7,8-HpCDF	3.30E-05		1.40E-04	
	1,2,3,4,7,8-HxCDF	3.74E-06		1.38E-05	
	OCDF	1.08E-04		4.47E-04	
ECO-10	2,3,7,8-TCDD	8.70E-08	JEMPC	1.34E-07	U
	1,2,3,7,8,9-HxCDD	1.84E-06	J	2.28E-06	U
	1,2,3,4,7,8,9-HpCDF	1.38E-06	J	1.80E-06	U
	1,2,3,6,7,8-HxCDD	2.60E-06		3.89E-06	J
	1,2,3,7,8-PCDD	3.01E-07	J	5.88E-07	J
	1,2,3,4,6,7,8-HpCDD	8.44E-05		1.25E-04	
	OCDD	7.39E-04		1.31E-03	
	2,3,7,8-TCDF	2.73E-07	U	9.79E-07	U
	1,2,3,7,8,9-HxCDF	6.19E-07		2.11E-07	U
	2,3,4,7,8-PCDF	3.30E-07	JEMPC	4.33E-07	JEMPC
	1,2,3,4,7,8-HxCDD	6.40E-07	J	7.99E-07	J
	1,2,3,6,7,8-HxCDF	6.73E-07	J	1.01E-06	JEMPC
	1,2,3,7,8-PCDF	4.54E-07	J	6.75E-07	J
	2,3,4,6,7,8-HxCDF	9.45E-07	J	1.41E-06	U
	1,2,3,4,6,7,8-HpCDF	1.60E-05		2.90E-05	
	1,2,3,4,7,8-HxCDF	1.71E-06	J	2.70E-06	U
	OCDF	4.63E-05		7.68E-05	

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-11	2,3,7,8-TCDD	2.27E-07	JEMPC	1.61E-07	U
	1,2,3,7,8,9-HxCDD	2.21E-05		8.65E-06	
	1,2,3,4,7,8,9-HpCDF	9.69E-06		2.87E-06	J
	1,2,3,6,7,8-HxCDD	3.46E-05		9.34E-06	
	1,2,3,7,8-PCDD	6.68E-06		1.74E-06	J
	1,2,3,4,6,7,8-HpCDD	5.44E-04		2.56E-04	
	OCDD	3.25E-03		1.81E-03	
	2,3,7,8-TCDF	3.38E-06		3.21E-06	U
	1,2,3,7,8,9-HxCDF	3.19E-06		1.01E-07	U
	2,3,4,7,8-PCDF	5.06E-06		1.09E-06	J
	1,2,3,4,7,8-HxCDD	1.19E-05		3.07E-06	JEMPC
	1,2,3,6,7,8-HxCDF	8.73E-07	J	1.76E-06	J
	1,2,3,7,8-PCDF	3.33E-06	EMPC	7.44E-07	J
	2,3,4,6,7,8-HxCDF	6.14E-06		1.34E-06	J
	1,2,3,4,6,7,8-HpCDF	9.50E-05		3.45E-05	
	1,2,3,4,7,8-HxCDF	1.34E-05		3.98E-06	J
	OCDF	3.17E-04		1.09E-04	
ECO-12[b]	2,3,7,8-TCDD	1.83E-06		1.28E-06	U
	1,2,3,7,8,9-HxCDD	2.35E-04		7.43E-05	
	1,2,3,4,7,8,9-HpCDF	8.68E-05		1.20E-04	
	1,2,3,6,7,8-HxCDD	1.94E-04		5.77E-04	
	1,2,3,7,8-PCDD	1.93E-05		1.69E-05	J
	1,2,3,4,6,7,8-HpCDD	7.44E-03		7.21E-03	
	OCDD	8.08E-02		4.37E-02	J
	2,3,7,8-TCDF	3.22E-06		2.88E-05	
	1,2,3,7,8,9-HxCDF	3.77E-05		5.01E-06	J
	2,3,4,7,8-PCDF	1.26E-05		8.32E-05	
	1,2,3,4,7,8-HxCDD	9.77E-05		2.72E-05	J
	1,2,3,6,7,8-HxCDF	2.16E-05		4.04E-05	
	1,2,3,7,8-PCDF	1.49E-05		5.13E-05	
	2,3,4,6,7,8-HxCDF	3.96E-05		3.61E-05	
	1,2,3,4,6,7,8-HpCDF	1.21E-03		1.19E-03	
	1,2,3,4,7,8-HxCDF	7.28E-05		1.61E-04	
	OCDF	4.25E-03		4.40E-03	

Notes

EMPC = estimated maximum possible concentration

J = estimated

U = not detected at the laboratory detection limit

a - Unless otherwise noted, earthworm tissue was collected at the same location as the adjacent soil data and has the same sample identification number.

b - Earthworm tissue is from a 28-d bioaccumulation test using Cass Lake soil.

Table D-2
Regression Relationships of Individual Dioxin and Furan Congeners in Colocated Cass Lake Soil and Earthworm Tissue

Congener	Relationship	Slope	Intercept	*p* value	R^2	Equation
2,3,7,8-TCDD	log-log	0.819	-2.494	**0.06**	0.53	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8-PCDD	log-log	0.516	-5.921	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,7,8-HxCDD	log-log	0.343	-7.481	0.2	0.16	ns
1,2,3,6,7,8-HxCDD	log-log	0.664	-3.420	**0.08**	0.48	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8,9-HxCDD	log-log	0.550	-5.043	**0.03**	0.65	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,6,7,8-HpCDD	log-log	0.479	-3.910	**0.1**	0.37	exp(intercept +slope*(ln($Cs_{congener}$))
OCDD	log-log	0.370	-3.424	0.2	0.26	ns
2,3,7,8-TCDF	log-log	0.251	-9.530	0.5	-0.12	ns
1,2,3,7,8-PCDF	log-log	0.593	-4.862	**0.08**	0.46	exp(intercept +slope*(ln($Cs_{congener}$))
2,3,4,7,8-PCDF	log-log	0.518	-5.916	0.2	0.29	ns
1,2,3,4,7,8-HxCDF	log-log	0.616	-4.292	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,6,7,8-HxCDF	log-log	0.609	-4.502	**0.03**	0.67	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8,9-HxCDF	log-log	0.671	-5.742	**0.07**	0.52	exp(intercept +slope*(ln($Cs_{congener}$))
2,3,4,6,7,8-HxCDF	log-log	0.576	-5.218	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,6,7,8-HpCDF	log-log	0.593	-3.688	**0.05**	0.56	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,7,8,9-HpCDF	log-log	0.453	-6.217	0.2	0.25	ns
OCDF	log-log	0.415	-4.735	0.2	0.28	ns

Notes

$Cs_{congener}$ = concentration of the given congener in soil

ns = no significant relationship

Regressions with $p \leq 0.1$ (in **bold)** are considered statistically significant and are used to construct regression equations.

Table D-3
Spearman's Correlation Coefficients (rho) for Dioxin and Furan Congeners in Cass Lake Earthworm Data For Which Regression Equations Could Not Be Developed

	1,2,3,7,8-PCDD	1,2,3,6,7,8-HxCDD	1,2,3,7,8,9-HxCDD	1,2,3,4,6,7,8-HpCDD	1,2,3,7,8-PCDF	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDF	1,2,3,4,6,7,8-HpCDF	2,3,4,6,7,8-HxCDF
1,2,3,4,7,8-HxCDD	**1.00**	0.96	**1.00**	0.93	0.96	**1.00**	**1.00**	0.82	**1.00**	0.96
OCDD	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93
2,3,7,8-TCDF	0.64	**0.71**	0.64	0.79	**0.71**	**0.64**	0.64	0.64	0.64	0.57
2,3,4,7,8-PCDF	0.96	**1.00**	0.96	0.96	**1.00**	0.96	0.96	0.86	0.96	0.93
1,2,3,4,7,8,9-HpCDF	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93
OCDF	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93

Notes

Coefficients in **bold** are significant ($p < 0.05$, see Table D-4), highly correlated,and have a significant regression equation (per Table D-2).

Table D-4
Significance (*p*- values) of Spearman's Correlation Coefficients for Dioxin and Furan Congeners for Which Regression Equations Could Not Be Developed

	1,2,3,7,8-PCDD	1,2,3,6,7,8-HxCDD	1,2,3,7,8,9-HxCDD	1,2,3,4,6,7,8-HpCDD	1,2,3,7,8-PCDF	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDF	1,2,3,4,6,7,8-HpCDF	2,3,4,6,7,8-HxCDF
1,2,3,4,7,8-HxCDD	0	8.7E-07	0	4.9E-04	8.7E-07	0	0	4.1E-01	0	7.9E-10
OCDD	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04
2,3,7,8-TCDF	2.0E-03	7.0E-03	2.0E-03	7.0E-01	7.0E-03	2.0E-03	2.0E-03	3.7E-03	2.0E-03	3.4E-04
2,3,4,7,8-PCDF	8.7E-07	0	8.7E-07	6.4E-04	0	8.7E-07	8.7E-07	1.4E-01	8.7E-07	2.4E-05
1,2,3,4,7,8,9-HpCDF	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04
OCDF	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04

Table D-5

Approach for Estimating Concentrations in Terrestrial Invertebrate Tissue of Congeners for Which a Significant Regression Equation Could Not Be Established

Congener of Interest	Related Congener	Average Ratio of Concentrations of the Congener of Interest to Related Congener Concentrations	CV
1,2,3,4,7,8-HxCDD	1,2,3,7,8-PCDD	1.94	0.221
	1,2,3,7,8,9-HxCDD	**0.430**	0.218
	1,2,3,4,7,8-HxCDF	0.375	0.575
	1,2,3,6,7,8-HxCDF	1.01	0.419
	1,2,3,4,6,7,8-HpCDF	0.0417	0.603
OCDD	1,2,3,7,8-PCDD	2,050	0.435
	1,2,3,7,8,9-HxCDD	460	0.443
	1,2,3,4,6,7,8-HpCDD	**8.02**	0.230
	1,2,3,4,7,8-HxCDF	349	0.407
	1,2,3,6,7,8-HxCDF	977	0.355
	1,2,3,4,6,7,8-HpCDF	38.0	0.340
2,3,7,8-TCDF	1,2,3,6,7,8-HxCDD[a]	**0.120**	1.179
	1,2,3,7,8-PCDF	1.16	1.398
	1,2,3,4,7,8-HxCDF[b]	**0.25**	1.228
2,3,4,7,8-PCDF	1,2,3,6,7,8-HxCDD	**0.108**	0.264
	1,2,3,7,8-PCDF	1.61	1.019
1,2,3,4,7,8,9-HpCDF	1,2,3,7,8-PCDD	5.06	0.768
	1,2,3,7,8,9-HxCDD	1.16	0.835
	1,2,3,4,6,7,8-HpCDD	0.0185	0.605
	1,2,3,4,6,7,8-HpCDF	0.0767	0.391
	1,2,3,4,7,8-HxCDF	**0.723**	0.332
	1,2,3,6,7,8-HxCDF	2.09	0.390
OCDF	1,2,3,7,8-PCDD	161	0.522
	1,2,3,7,8,9-HxCDD	36.2	0.546
	1,2,3,4,6,7,8-HpCDD	**0.603**	0.215
	1,2,3,4,7,8-HxCDF	25.1	0.257
	1,2,3,6,7,8-HxCDF	72.6	0.334
	1,2,3,4,6,7,8-HpCDF	2.79	0.256

Notes

CV = Coefficient of variation, a measure of dispersion of the data, which is calculated as the standard deviation of the ratios of the congener to its correlate divided by the average ratio.

Selected average ratio for estimating tissue concentrations for the congener of interest are in **bold**.

a - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples outside of the impoundments.

b - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples inside the impoundments.

Table D-6
Estimated Concentrations of Dioxins and Furans (TEQ_{DF}) in Terrestrial Invertebrate Tissue at the Site

	CT, mg/kg dw	RM, mg/kg dw
With RBA[a]		
$TEQ_{DF,B}$[b]	6.07E-05	1.81E-04
$TEQ_{DF,M}$[c]	9.42E-05	2.84E-04
Without RBA[a]		
$TEQ_{DF,B}$[b]	1.17E-04	3.59E-04

Notes

CT = central tendency

RM = reasonable maximum

a - Relative bioavailability adjustment factor applied to TCDD congener for calculating $TEQ_{DF,B}$

b - Calculated using soils north of I-10, consistent with the killdeer exposure scenario

c - Calculated using peninsula-wide soils, consistent with the raccoon exposure scenario

FIGURES

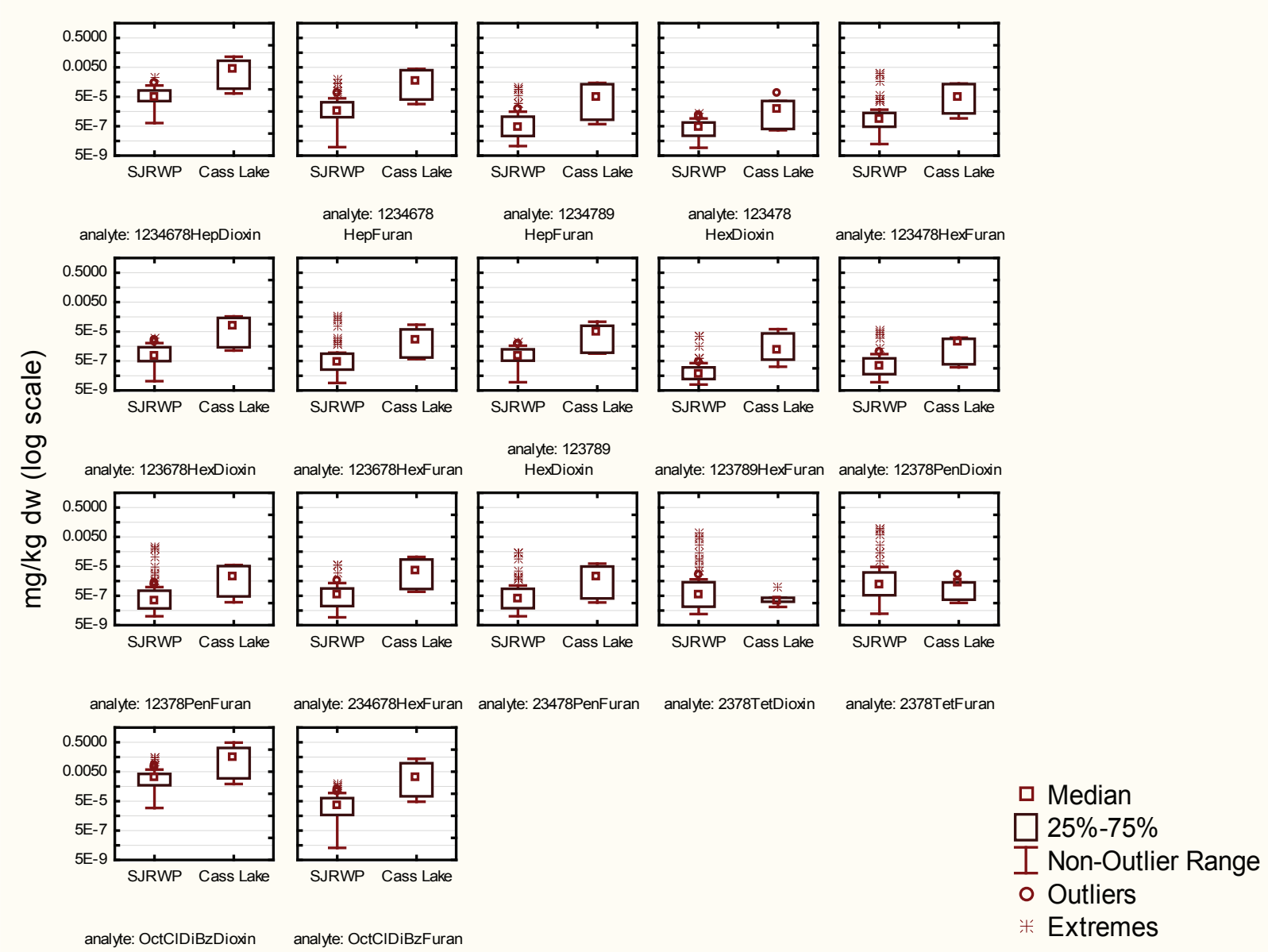

Figure D-1
Box Plots of Dioxin and Furan Congener Concentrations in Cass Lake and all SJRWP Soils from North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

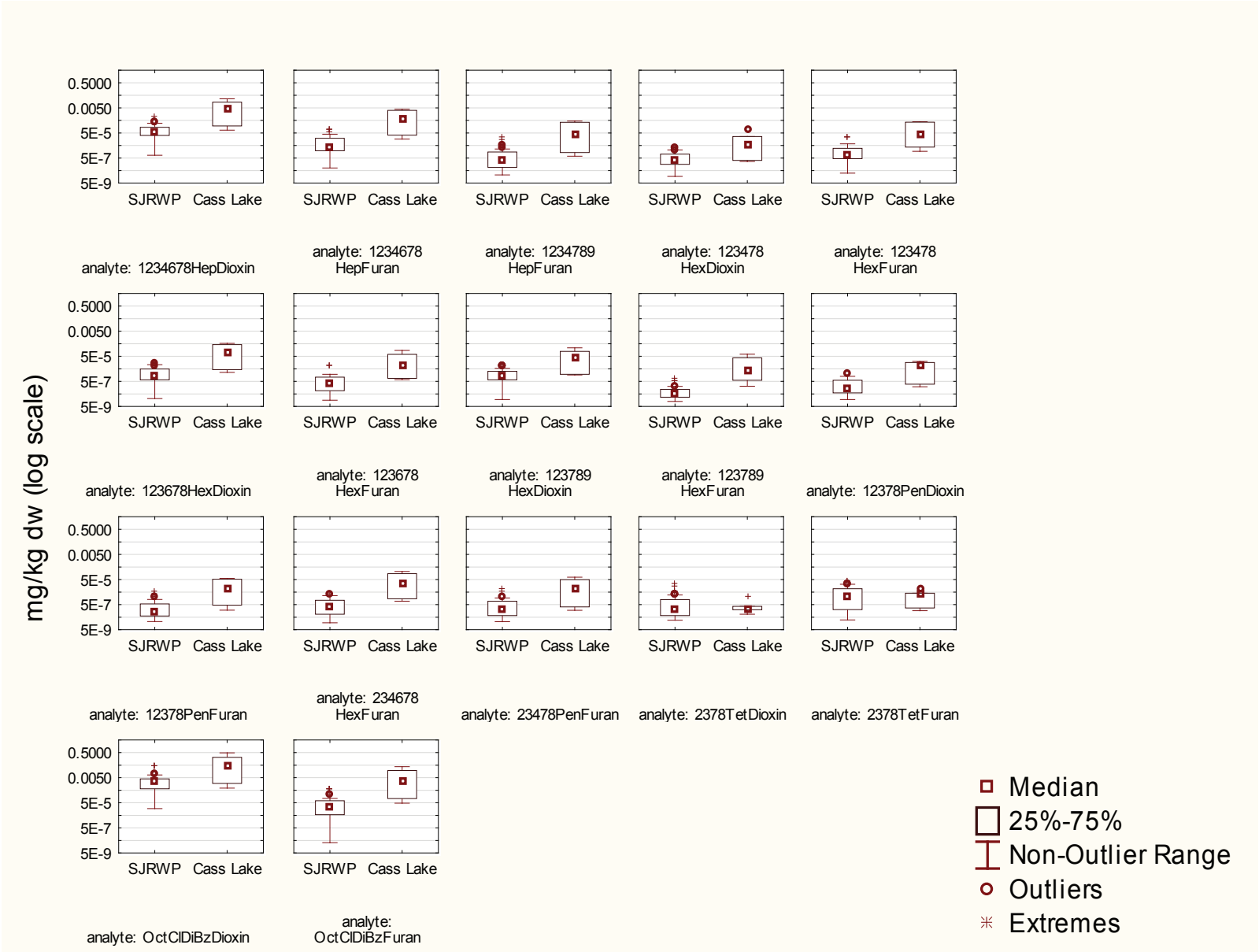

Figure D-2
Box Plots of Dioxin and Furan Congener Concentrations in
Cass Lake and SJRWP Soils Outside of the Waste Impoundments North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

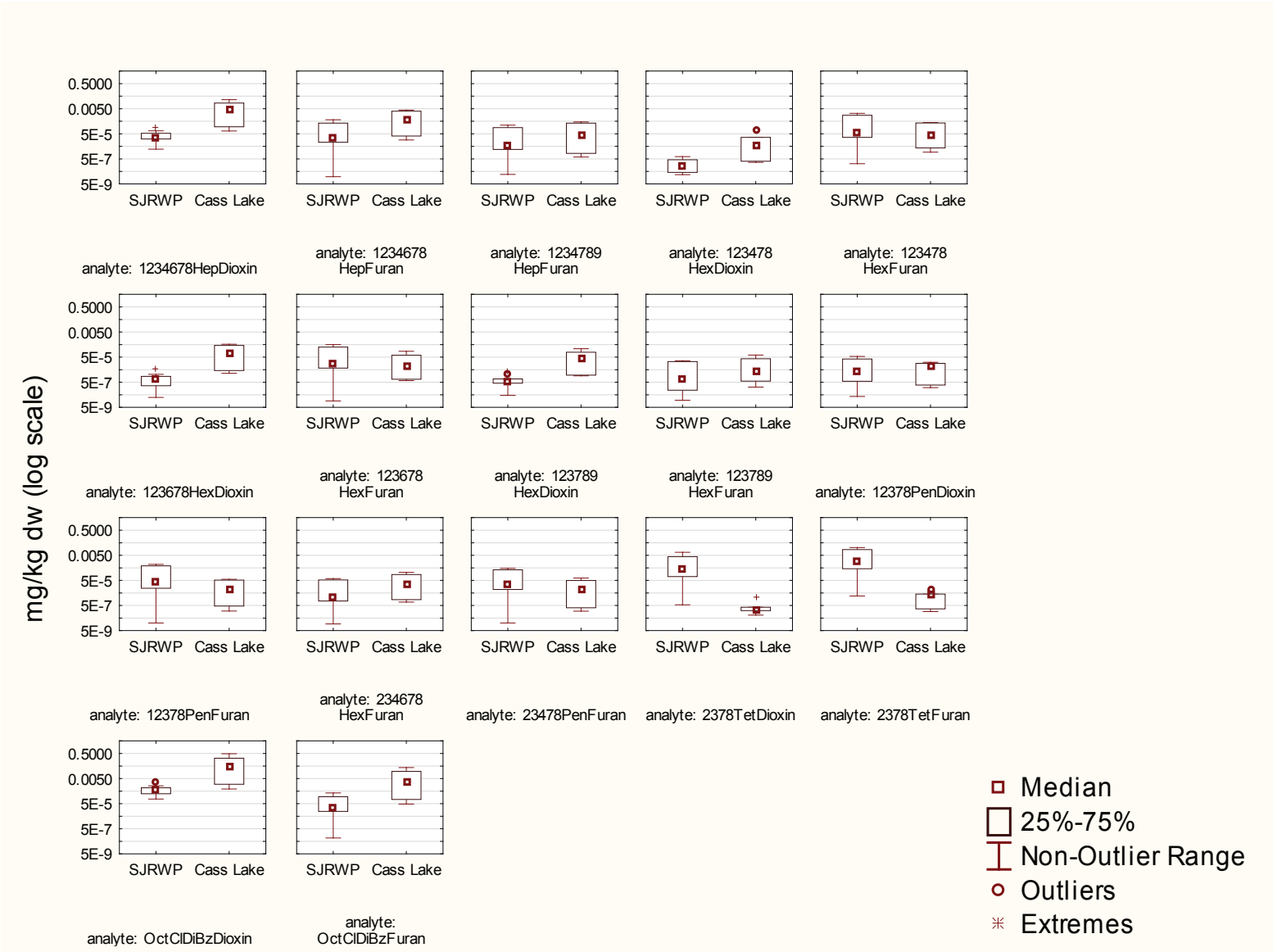

Figure D-3
Box Plots of Dioxin and Furan Congener Concentrations in
Cass Lake and SJRWP Soils Inside of the Waste Impoundments North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

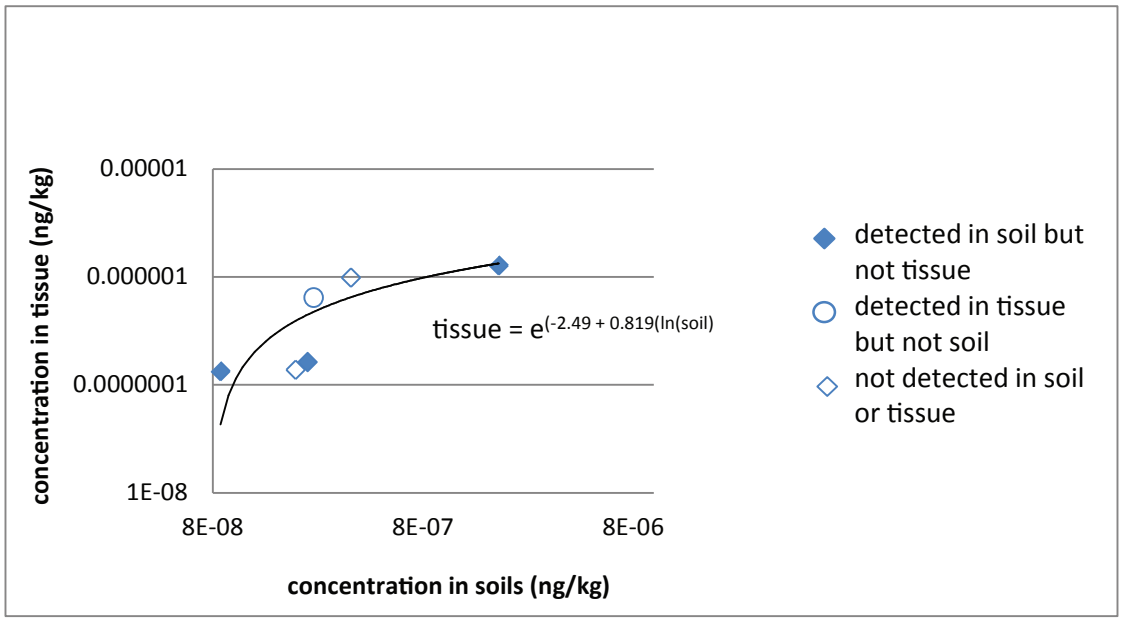

Figure D-4
Relationship of 2,3,7,8-TCDD in Cass Lake Colocated Soil and Earthworm Tissue Samples
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

APPENDIX E
SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT, SOUTH IMPOUNDMENT

SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

SOUTH IMPOUNDMENT

SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

International Paper Company

U.S. Environmental Protection Agency, Region 6

Prepared by

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
CDD	chlorinated dibenzo-*p*-dioxins
COI	chemical of interest
COPC	chemical of potential concern
$COPC_E$	chemical of potential ecological concern
CSM	conceptual site model
DQO	data quality objective
EcoSSL	ecological soil screening level
ERA	ecological risk assessment
I-10	Interstate Highway 10
IPC	International Paper Company
K_{ow}	octanol-water partition coefficient
PCB	polychlorinated biphenyl
RI/FS	Remedial Investigation and Feasibility Study
SAP	sampling and analysis plan
Site	San Jacinto River Waste Pits Site
SLERA	screening-level ecological risk assessment
SLV	screening level value
SMDP	scientific management decision point
SVOC	semivolatile organic compound
TCEQ	Texas Commission on Environmental Quality
USEPA	U.S. Environmental Protection Agency

1 INTRODUCTION

This Screening-Level Ecological Risk Assessment (SLERA) for the south impoundment area of the San Jacinto River Waste Pits site (the Site) has been prepared on behalf of International Paper Company (IPC), pursuant to the requirements of Unilateral Administrative Order, Docket No. 06-03-10, which was issued by the U.S. Environmental Protection Agency (USEPA) to IPC and McGinnes Industrial Maintenance Corporation on November 20, 2009 (USEPA 2009). This SLERA presents information to supplement the SLERA prepared as Appendix B to the Remedial Investigation/Feasibility Study (RI/FS) Work Plan (Anchor QEA and Integral 2010).

This document is submitted as Appendix E to the Baseline Ecological Risk Assessment (BERA), and uses results from the Phase I soil investigation conducted in March 2011 in Soil Investigation Area 4 (Integral 2011b). Results of the Phase I investigation for this area have been presented in the Preliminary Site Characterization Report (PSCR) (Integral and Anchor QEA 2012). Analyses of the data according to the data quality objectives (DQOs) of the Phase I study are presented in Attachment A to the draft Soil Sampling and Analysis Plan (SAP) Addendum 3 (Integral 2011a). This SLERA was prepared consistent with USEPA guidance for ecological risk assessment (USEPA 1998) and addresses Step 1 and Step 2 of the 8-step ecological risk assessment (ERA) process for Superfund (USEPA 1997) (Figure E-1). Preparation of a SLERA for the south impoundment provides the basis for a BERA for the south impoundment, which will be submitted with the Remedial Investigation Report. The south impoundment BERA will be conducted using the approach and methods for completing ERA Steps 3 through 8 as described in the RI/FS Work Plan.

This SLERA provides the screening-level problem formulation and ecological effects evaluation (Step 1) and the screening-level exposure assessment and risk evaluation (Step 2), either as a unique section in this document, or by reference to Appendix B of the RI/FS Work Plan. Several components of the SLERA are addressed only briefly in this document because they are described in greater detail elsewhere:

- Site history and facilities used at the Site (Section 2 of the RI/FS Work Plan and Section 1.4.1 of Soil Sampling and Analysis Plan (SAP) Addendum 1 (Integral 2011b)
- Identification of chemicals of interest (COIs) (Appendix C to the RI/FS Work Plan)

- Complete evaluation of Phase I soil investigation results according to DQOs in Attachment A to Soil SAP Addendum 3 (Integral 2011a).

This SLERA is intended to provide a description of the environmental setting of Soil Investigation Area 4, and to document the scientific management decision points (SMDPs) to transition from a general understanding of the south impoundment environment to the more Site-specific study design elements and analyses required for performing the BERA. The SLERA is organized as follows:

- **Section 2**. Screening-Level Problem Formulation. This section reviews the relevant information for the south impoundment area. It includes by reference related information from the overall SLERA for the Site. The result is a list of receptor surrogates and an ecological conceptual site model (CSM) for the south impoundment area.
- **Section 3**. Screening-Level Evaluation and Identification of Chemicals of Potential Ecological Concern (COPCEs). This section describes the basis for the screening-level values (SLVs) used in the risk-based screens, and presents a screening analysis of data collected during the initial soil investigation described by Soil SAP Addendum 1 (Integral 2011b), and results in the identification of COPCEs.
- **Section 4**. Scientific Management Decision Point (SMDP). A summary of the findings of the SLERA is presented as an SMDP for the south impoundment ERA process. Uncertainties in the analyses are discussed.

2 SCREENING LEVEL PROBLEM FORMULATION

The screening level problem formulation uses existing information to develop a preliminary CSM for ecological receptors that addresses the following (USEPA 1997):

- The environmental setting and contaminants known or suspected to occur
- Mechanisms of contaminant fate and transfer
- Mechanisms of toxicity and likely categories of receptors that could be affected
- The complete exposure pathways linking contaminants to ecological receptors in the south impoundment environment
- Endpoints that can be used to screen for potential ecological risk.

This section summarizes basic information on the environmental setting, chemical fate and transport mechanisms relevant to developing the CSM, receptors potentially in the south impoundment area of the Site, and the surrogates to be used for risk assessment, and it defines the assessment endpoints for the screening level analysis. A detailed discussion of the toxicity of dioxins and furans is provided in Attachment B2 to Appendix B of the RI/FS Work Plan. The resulting CSM synthesizes this preliminary assessment to identify mechanisms of exposure and effects that may result in contaminant-related risks to ecological receptors. The CSM may be refined in the problem formulation presented in the BERA to better reflect site-specific exposures and risks.

2.1 Environmental Setting

Several historical aerial images of this area of the Site were analyzed to determine the location and history of the impoundment south of Interstate 10 (I-10). Review of historical documents and aerial photographs indicates that an impoundment (Figure E-2) was constructed in the mid-1960s south of I-10, on the peninsula directly south of the northern impoundments on the western bank of the San Jacinto River, in Harris County, Texas (this area is Soil Investigation Area 4 for the purposes of the RI/FS). This southern impoundment was used for disposal of paper mill wastes. Beginning in the 1970s, much of the peninsula south of I-10 underwent substantial physical change due to road, parking lot, and building development. Additional description of the possible configurations of the original southern impoundment, documented historical waste disposal and characteristics of that waste, and physical changes at the Site from 1957 to the present are provided by Integral (2011b).

The upland areas south of I-10 are currently under industrial or commercial use, including use by a shipbuilding company and an active shipyard. The shipyard has been operational since 1957, and the shipyard property was the site of a waste impoundment used for management of wastes associated with barge repair and cleaning materials (e.g., grinding or blasting wastes and cleaning solutions). A recent application for a municipal settings designation by the shipyards (W&M 2011) describes an ongoing groundwater treatment program, and the history of groundwater monitoring and remediation on the shipyards property, which is to the west of Soil Investigation Area 4.

The PSCR (Integral and Anchor QEA 2012) provides a more complete summary of recent information on commercial and industrial activities in the vicinity of the south impoundment, and uses observations made during the Phase I soil investigation to update the south impoundment CSM that was developed in Soil SAP Addendum 1 (Integral 2011b). The most recent CSM, from the final PSCR, is presented in greater detail in Section 2.6.

2.1.1 Habitat

In natural low-elevation habitats adjacent to the San Jacinto River, soils consist primarily of clay and sand and support loblolly pine-sweetgum, loblolly pine-shortleaf pine, water oak-elm, pecan-elm, and willow oak-blackgum (TSHA 2009). The area of the south impoundment has been cleared and graded for industrial use. This area is generally flat with very little topographic relief, and a small rise in elevation from 6 feet above sea level in the north to greater than 10 feet above sea level at the southern end of the impoundment. Most of the area is covered with mowed grasses and forbs and there are few trees throughout the Site. Shorelines in the area are covered by rip-rap, and fringe wetlands are populated by phragmites, small trees, and shrubs, particularly along the northern boundary of the south impoundment. Shallow estuarine waters abut the shoreline, and deeper estuarine waters offshore to the west of the Site are maintained for shipping activities.

2.1.2 Contaminants Known or Suspected to Occur at the Site

The process to identify COIs (Table E-1) for the Site is described in Appendix C of the RI/FS Work Plan (Anchor QEA and Integral 2010). COIs were defined as those chemicals that are

among USEPA's priority pollutants, were reported by one or more technical papers as potentially occurring in pulp mill solid wastes or leachate from solid waste landfills containing pulp mill wastes, and are likely to have bound to organic carbon or could otherwise have persisted for more than 40 years in the Site environment. The Phase I soil investigation for the Site generated information on the spatial distribution and concentrations of COIs in soils of the south impoundment area (Integral 2011a).

COPCEs for the south impoundment area have not been identified. The methods to determine COPCEs, and the analyses of Phase I soil data to identify COPCEs for evaluation in the BERA is presented below, in Section 3.

2.2 Contaminant Fate and Transport

The characterization of contaminant fate and transport includes identification of 1) pathways for migration of COIs at the Site; and 2) physical, chemical, and biological transformations of these COIs. Understanding general mechanisms of fate and transport helps define the exposure pathways to ecological receptors that may be adversely affected by Site contaminants (USEPA 1998). This section provides general information on chemical transport and transformation pathways for COIs in soils of the south impoundment on the basis of currently available information. The PSCR provides a more detailed discussion.

2.2.1 Physical Fate and Transport Processes

Potential transport pathways for COIs in south impoundment soils include transport of contaminated soils via surface runoff, and transport of dissolved COIs via groundwater to surface water. The updated CSM for the south impoundment presented in the PSCR (Figure E-3) differentiates surface soil and subsurface soil on the basis of information developed in the Phase I soil investigation. If surface soils in the south impoundment area are affected by paper mill wastes, and surface water runoff pathways in the area could transport soils to the Old River, sediment and water could be affected by COIs originating in the south impoundment.

To evaluate the potential for physical transport of COIs from the surface soils to the aquatic environment, the topography and surface water flow paths south of I-10 were described in

the PSCR (Section 7.1). Surface water flow paths were shown to comingle into larger drainage networks, but to ultimately either discharge to the river on either side of the peninsula or terminate in surface depressions, at which surface water runoff would be expected to aggregate and ultimately percolate into the soil. Given that dioxin and furan concentrations in the three sediment samples nearest the south impoundment are below the reference envelope value for soil, and that the concentrations of dioxins and furans in the majority of surface soil samples were also below concentrations in background soils, the potential for surface transport of soils affected by paper mill wastes to the aquatic environment is considered negligible (Integral and Anchor QEA 2012). USEPA has requested that additional information on sediments and soils be collected as part of a subsequent phase of investigation.

The subsurface soils 6 feet and deeper below ground surface may be in contact with groundwater, and chemicals could be transported in a dissolved state or bound to particulates from the subsurface soil environment to the aquatic environment. The groundwater study conducted north of I-10 (described in Section 6 of the PSCR) is relevant to the CSM for the south impoundment because it demonstrated that, even in an area where there are concentrated wastes situated in alluvial sediments, contamination of alluvial groundwater and the deep aquifer with dioxins and furans did not occur. These results suggest that in the vicinity of the south impoundment, where the data indicate that paper mill wastes are substantially less concentrated than in the location of the groundwater study, that there may also be a very limited or no groundwater pathway resulting in the transport of dioxins and furans to receptors. Also, dioxins and furans strongly adsorb to soil particles and have very low solubility and mobility in groundwater (Fan et al. 2006; USAF 2006; ATSDR 1998). ATSDR (1998) indicates that chlorinated dibenzo-*p*-dioxins (CDDs) "…bind strongly to the soil, and therefore are not likely to contaminate groundwater…" and "CDDs are unlikely to leach to underlying groundwater…" These properties further decrease the likelihood that dioxins and furans are transported by groundwater from the subsurface soils to the aquatic environment.

However, USEPA contends that additional groundwater data are necessary to demonstrate that conditions observed north of I-10 are representative of those south of I-10, and has requested that information on the chemistry of alluvial groundwater in the area south of I-10

be collected. If there is a groundwater pathway resulting in contamination of the aquatic environment with chemicals from the southern impoundment, analysis of the sediment and tissue data for the overall Site in the BERA will address any related ecological risks.

2.2.2 Biological Fate and Transport Processes: Bioaccumulation

Bioaccumulation is relevant to the BERA for several chemicals. A simple definition of bioaccumulation is the sequestration of a chemical substance in an organism when the absorption rate (from exposures to all media) exceeds the elimination or transformation rates, resulting in the concentration in tissue exceeding the concentration in the exposure medium. Bioaccumulation dynamics and rates are specific to the substance of concern, the exposure route, the medium or media in which the chemical is delivered, and the type of organism. Biomagnification is related to bioaccumulation and describes the increase in the concentration of a substance with increasing trophic level in a food chain (e.g., from primary to tertiary consumer). Biomagnification appears to be restricted to a relatively small group of chemicals (Croteau et al. 2005; Suedel et al. 1994).

A key indicator of the potential for bioaccumulation is the chemical's hydrophobicity, which is most often expressed using the *n*-octanol–water partition coefficient, K_{ow}, and has been used to predict bioaccumulation potential. Hydrophobic and lipophilic organic compounds that are resistant to both degradation and excretion in organisms build up in adipose tissue. Generally, organic chemicals that significantly bioaccumulate are those that are non-ionic, have a log K_{ow} of 5 or greater, and are not rapidly metabolized or excreted (USEPA 2008a). More recent literature indicates that dioxin and furan congeners that are not tetrachlorinated at the 2, 3, 7, and 8 positions have very limited bioaccumulation potential in vertebrates (USEPA 2008b). The bioaccumulation of dioxins and furans is addressed by Integral (2010).

Metals bioaccumulation is complex, and bioaccumulation rates can vary with the concentration in the exposure medium. As a result, simple models of metals bioaccumulation, such as the use of bioaccumulation factors, may lead to inaccurate depiction of concentrations in tissue (USEPA 2007). Bioaccumulation is considered a relevant process for determining the fate of COIs at the Site, and chemical-specific bioaccumulation potential based on the Texas Commission on Environmental Quality

(TCEQ) guidance (TCEQ 2006) is incorporated into the risk-based screens that are applied in Appendix C (to the RI/FS Work Plan) to identify chemicals of potential concern (COPCs) for ecological (and human) receptors. The findings of the Technical Memorandum on Bioaccumulation Modeling (Integral 2011d) including the conceptual framework on bioaccumulation of dioxins and furans, and the general conclusions regarding the appropriate models for predicting tissue concentrations, and the chemicals considered bioaccumulative by TCEQ (2006) provide the basis for consideration of trophic transfer to ecological receptors for this risk assessment.

2.3 Selection of Surrogate Ecological Receptors

This section builds from Appendix B and Attachment B1 to the RI/FS Work Plan (Anchor QEA and Integral 2010) to describe the ecological receptors that could occur in the vicinity of the south impoundment at the Site, identify surrogate species to be evaluated in the BERA for the south impoundment and present the rationale for their selection. A surrogate receptor species is chosen to represent a group of related species with similar feeding patterns, habitat associations, or other life history characteristics that affect the exposure potential of the receptor group.

2.3.1 Selected Receptor Surrogates

Ecological receptor surrogates are considered representative of the trophic and ecological relationships known or expected at the Site. In selecting receptor surrogates for evaluation in the BERA for the Site, the following criteria were considered:

- The receptor is or could potentially be present at the Site
- The receptor is representative of one or more feeding guilds
- The receptor is known to be either sensitive or potentially highly exposed to COPCs at the Site
- Life history information is available in the literature or is available for a similar species that can be used to inform life history parameters for the receptor.

Detailed tables listing the species of plants, reptiles, birds, and mammals that could use the upland habitats on the Site or in the vicinity of the Site are provided in the SLERA for the overall Site, as Attachment B1 to the RI/FS Work Plan (Anchor QEA and Integral 2010).

Using the guidelines listed above, four upland receptors (a bird, two mammals, and a reptile) were selected for evaluation in the BERA for the southern impoundment (Table E-2). Additional information on these receptors' life history and feeding behavior is provided below.

2.3.1.1 Reptile—Common Garter Snake

The garter snake was selected because it is a common, invertivorous reptile whose habitat requirements overlap with the conditions present in the upland portions of the Site.

The common garter snake (*Thamnophis sirtalis*) is one of the most abundant snakes in North America. Of the four subspecies of the common garter snake found in Texas, the Texas garter snake (*Thamnophis sirtalis annectens)* is the only subspecies known to inhabit eastern Texas locations; Harris County is one of several upper Gulf Coast counties in which these snakes have been observed in the last decade (Cannatella and LaDuc 2011). Regional populations of common garter snakes across the continent are distinguished mostly by variation in color patterns. The adult common garter snakes range in size between 46 and 137 cm (18 and 54 inches), and weigh an average of 150 g. The males are smaller than females and the young, which are similar in appearance to the adults, are 12.5 to 23 cm (5 to 9 inches) long at birth (Zimmerman 2002).

The adaptability and resilience of the common garter snakes are evidenced by their residence in a wide variety of terrestrial and semiaquatic habitats, including meadows, marshes, woodlands, hillsides, and suburban and urban areas where debris, rock walls, foundations, gardens and other features provide good cover. These snakes prefer moist, grassy environments such as is found near the edges of ditches, ponds, lakes and streams (Zimmerman 2002). In Texas, these snakes are found primarily in lowland habitats, particularly in areas with standing or running water, but can also be seen in open or edge habitats (Cannatella and LaDuc 2011). Similar to most reptiles, the common garter snake uses thermoregulation to achieve a preferred body temperature between 28 and 32°C. While these snakes tolerate a broader range of temperatures than do most, they bask in the sun during the day, and convene in coiled masses during sleep or hibernation to retain body heat. Hibernation occurs in natural cavities, rodent or crayfish burrows, under rock piles, or in stumps.

The common garter snake eats a variety of prey, dependent primarily on whether it is appropriately sized for swallowing whole. The adult diet includes amphibians, fish, and insects. Juvenile garter snakes eat a greater proportion of earthworms and insects than do adults. Baby birds, mammals, molluscs, and other snakes are also taken as prey items (Cannatella and LaDuc 2011)

Garter snakes mate in the spring, as soon as they emerge from hibernation, and are ovoviviparous, meaning they carry their young until birth. In the summer and fall, the females birth an average of 26 young. The mother snakes allow the young to be around them for several days after birth, but do not provide any care, protection, or nourishment. These snakes reach sexual maturity, and maximum size, at 3 to 4 years of age, though Zimmerman (2002) indicates that the average lifespan of common garter snakes is approximately 2 years and that most common garter snakes probably die in their first year of life.

Common garter snakes are eaten by a wide variety of predators, including large fish, bullfrogs, snapping turtles, milk snakes, American crows, hawks, great blue herons, raccoons, foxes, squirrels, and shrews.

2.3.1.2 Bird—Killdeer

The killdeer (*Charadrius vociferous*) was selected because it is an upland bird whose habitat requirements overlap with the conditions present in the upland portions of the site. Only terrestrial birds are expected to be present in the south impoundment area.

The killdeer is an upland plover that feeds mainly on terrestrial invertebrates (e.g., earthworms, beetles, grasshoppers, and other small invertebrates). Stomach contents from killdeer in Texas were reported to contain 98 percent animal matter, mostly worms and insects (McAtee and Beal 1924). The species is widespread throughout North America, using open area habitats (e.g., agricultural fields, lawns, golf courses). The killdeer is non-migratory within its range in the southern United States, including Texas (Jackson and Jackson 2000). It is known to be common year-round in the vicinity of the Site (Attachment B1 to the RI/FS work plan [Anchor QEA and Integral 2010]). This species is tolerant of constructed

disturbances, and nesting has been documented to occur in construction sites, road shoulders, and graveled rooftops (Jackson and Jackson 2000). Average nesting home ranges of killdeers in Minnesota were relatively small (0.57 acres). Larger, year-round home ranges of approximately 15 acres are reported elsewhere; nesting period home ranges are smaller. Nesting in Mississippi occurs from mid-March through late July and involves multiple broods (Jackson and Jackson 2000).

Due to its likely presence in the upland portions of Site, its relatively small home range and site fidelity, and its predominantly terrestrial invertebrate diet, the killdeer is representative of the species that would be subject to ecological risks associated with the terrestrial food chain at the Site. The use of this surrogate species is considered protective of smaller home range bird species at the Site (e.g., sparrows, wrens) that likely eat a larger percentage of plant matter, as well as larger omnivores (e.g., crows), and would also be protective of terrestrial carnivores (e.g., hawks) that likely have larger home and forage ranges.

2.3.1.3 Mammal—Pocket Gopher

The Baird's pocket gopher (*Geomys breviceps*), also known as the Louisiana pocket gopher, is virtually indistinguishable, morphologically, from the plains (*G. busarius*) and Attwater's (*G. attwateri)* pocket gophers, each of which inhabit different regions of Texas (Sulentich et al. 1991; Davis and Schmidly 1994). These pocket gophers are small, dark brown, burrowing herbivores. With long, curved and specially adapted front claws, a broad, flat head, tiny, bead-like eyes and rudimentary ears, and a compact body with skin and hair arranged to allow movement through borrows both backward and forward, these gophers are more highly specialized for digging than any other North American rodent (Davis and Schmidly 1994; KSR 2011; Sulentich et al. 1991). *G. breviceps* is the smallest of its congenerics, averaging 208 mm in length and weighing between 78 and 150 g, with an average reported weight of 100 g (MNH 2012). The Baird's pocket gopher is found in the eastern portion of Texas and has been found on both sides of the San Jacinto River in Harris County (Sulentich et al. 1991; Davis and Schmidly 1994).

Geomys live underground most of their lives and maintain labyrinths of burrows in sandy and loamy soils, digging to an average depth of approximately 6 inches and up to 2 feet,

generally on treeless land (Davis and Schmidly 1994). Because much of the burrowing is done in search of food, tunnels meander through feeding areas, and can extend well over 100 m. These rodents are solitary; each tunnel system is occupied by only one gopher. They rarely leave their burrows, except at night for mating or for limited foraging beyond the entrance (KSR 2011). In wet months, pocket gophers are known to live and nest in above-ground mounds of dirt, in order to avoid being flooded out of their burrows and tunnels (Sulentich et al. 1991).

The Baird's pocket gopher is an herbivore, obtaining most of its food while digging tunnels and feeding primarily on underground roots and the stems of weeds and grasses. Although most plant food is encountered and ingested while the gopher digs its lateral tunnels, green plants and grasses are obtained at night from around the entrance of the tunnels and beyond. Fur-lined cheek pouches are used to carry food and nesting material. Cellulose-digesting bacteria in the digestive system help the Baird's pocket gopher digest grasses and stored underground rhizomes during the winter and these gophers, as do many rodents, increase their utilization of food by ingesting their fecal pellets (Sulentich et al. 1991; Davis and Schmidly 1994).

The Baird's pocket gopher begins breeding in eastern Texas in early February and continues through August, with peak productivity occurring in June and July. One to four young are born to each litter (Sulentich et al. 1991). The young remain with their mother until nearly full-grown, at about 6 to 7 weeks of age, when they disperse to lead an independent life (Davis and Schmidly 1994). Sexual maturity is reached within 90 days of birth (Sulentich et al. 1991).

In east Texas, Baird's pocket gophers are preyed on by long-tailed weasels, and, when caught out of their burrows, are vulnerable to king snakes, great-horned owls, red-tailed hawks, and striped skunks, among other common rodent predators (Sulentich et al. 1991; Davis and Schmidly 1994). Because they remain protected in their borrows most of the time, pocket gophers are long-lived relative to many other rodents, living an average of 1 to 2 years in the wild (Davis and Schmidly 1994). The estimated population density in prairie habitat near College Station, Texas, was approximately 0.55 gophers per hectare (Sulentich et al. 1991).

2.3.1.4 Mammal–Virginia Opossum

The Virginia opossum (*Didelphis virginiana*) was selected because it is an omnivorous mammal whose habitat requirements overlap with the conditions present in the upland portions of the site.

The Virginia opossum is a widespread and adaptable nocturnal scavenger similar in size to a large house cat (UMMZ 2003). It is the only marsupial found north of Mexico. Opossums range from Central America through much of the continental United States, including the eastern two-thirds of the country and the coastal Pacific. Opossums range in size from 350 to 940 mm, averaging 740 mm. Adult males weigh an average of 5.5 pounds, and adult females average 4.0 pounds (Georgia DNR 2012); size may vary with location and climate (MNH 2012). The lifespan of a Virginia opossum averages 2 years, though many die in the first year of life (TPWD 2012). Both northern and southern populations have white fur with black tips. They have a pointed snout, opposable thumb-like appendages and a scaly prehensile tail that can be used to climb, hang, or grasp objects (TPWD 2012).

Opossums are well adapted to living near humans and occur in a variety of habitat types. They are primarily found in woodland areas especially near creeks, rivers, or lakes, but can also occupy marshes, farmland, prairies, and urban and rural environments. They prefer to live in hollow trees and logs, but can also nest under rocks, buildings, bridges, attics, woodpiles or in other animals' abandoned burrows. (UMMZ 2003; Georgia DNR 2012). In East Texas, Virginia opossums typically frequent overlapping home ranges approximately 0.05 km^2 in size, although the minimum size of home ranges may vary from 0.001 to 0.23 km^2. In East Texas woodland habitat, the density of opossums is about one opossum every 0.02 km^2 while in sandy, coastal parts of the state, the density is about one opossum every 0.06 km^2 (Davis and Schmidly 1994).

The Virginia opossum has a brief gestation period of 2 weeks after which the relatively undeveloped young crawl from the birth canal and attach themselves to the mother's nipple inside of her fur-lined pouch, where they stay attached for 7 weeks of nursing (UMMZ 2003). Litters usually consist of seven young, and Virginia opossums typically have two litters per year (Georgia DNR 2012).

Virginia opossums are omnivorous. Consuming mostly insects and carrion, the opossum also forages for acorns, berries, and other fruit and is also known to eat crustaceans, frogs, bird eggs and nestlings, small rodents, and the young of its own kind. In human-populated areas the opossum is known to scavenge for garbage and can be considered a nuisance for this reason (Georgia DNR 2012).

Common predators of Virginia opossums include canids, raccoons, and raptors. Humans are also a main cause of mortality through hunting and trapping, and opossums are frequently killed on roads (Georgia DNR 2012). Opossums are considered a game animal and in many states there are rules and regulations pertaining to their harvest through trapping and hunting. Despite their appeal to hunters, biologists do not believe that hunting is a threat to most populations of this species (Georgia DNR 2012).

2.3.2 Threatened and Endangered Species

Attachment B1 to the RI/FS work plan (Anchor QEA and Integral 2010) provides lists of species that could occur at the Site. Among the animals listed in Attachment B1 to the RI/FS work plan (Anchor QEA and Integral 2010), the upland species that are state-listed as threatened or endangered are:

- Timber rattlesnake
- Smooth green snake
- Rafinesque's big-eared bat.

The two snakes that are listed above have habitat requirements that are inconsistent with conditions present on the site (Anchor QEA and Integral 2010). The common garter snake has been selected as the surrogate receptor for reptiles at the Site. The Rafinesque's big-eared bat is not expected to use the habitats found in the vicinity of the Site because it feeds primarily on emergent aquatic insects, which are generally restricted to freshwater systems and are uncommon in brackish estuarine waters found near the Site.

In addition to these listed species, the American bald eagle, protected under the federal Bald and Golden Eagle Protection Act and listed as threatened by the State of Texas may be found in the vicinity. The American bald eagle may hunt for fish or eat carrion found on terrestrial

and shoreline areas. Given the limited size and habitat south of I-10, the bald eagle is considered unlikely to occur and is not addressed specifically for Soil Investigation Area 4.

2.4 Potential Routes of Exposure

For an exposure pathway to be complete, a contaminant must be able to travel from its source to an ecological receptor, and to be taken up by the receptor by one or more exposure routes. Complete exposure pathways for terrestrial wildlife result from ingestion of contaminated soil; ingestion of prey organisms that have been exposed to contaminated media and have bioaccumulated COIs; direct contact with contaminated soil; and inhalation of volatile chemicals in confined spaces (burrows). Interpretation of the significance of each exposure route in any species is dependent upon the availability of information in the literature. This section describes in general terms the routes of exposure of ecological receptors to chemicals on the south impoundment portion of the Site. This information provides the basis for the CSM for ecological exposures in the area of the south impoundment.

2.4.1 Ingestion

Direct ingestion of chemicals is commonly used to evaluate exposure in an ERA because much of the available and relevant toxicity literature for birds and mammals reports on the oral toxicity of chemicals and because many receptors ingest multiple contaminated media (i.e., food, water, and soil). The oral dose is considered greatest among the possible exposure routes for most terrestrial species.

Reptiles ingest soil directly while burrowing or foraging. Birds and mammals can ingest soil directly while foraging and cleaning their fur or feathers (Beyer et al. 1994). Reptiles, birds, and mammals also ingest bioaccumulative COIs through consumption of contaminated prey tissue. The extent to which trophic transfer via ingestion occurs is dependent on numerous factors, including the exposure of the prey to COIs, the bioaccumulation potential of the specific chemical, the extent to which the chemical is partitioned in the tissues of the prey, and what parts of the prey are eaten by the receptor. Trophic transfer is of particular importance for hydrophobic bioaccumulative chemicals of concern and for higher trophic-level consumers (e.g., raptors and carnivorous mammals).

2.4.2 Direct Contact

For terrestrial ecological receptors, direct contact exposure may include uptake across the integument (an enveloping layer such as a skin, membrane, or cuticle). The extent of direct contact with the exposure medium depends on the chemical and the physiology, habitat, and life history characteristics of the ecological receptor. Although direct contact exposure via transfer across external tissues is possible in ecological receptors, it is rarely quantified directly in ERAs because data are not available in the literature to interpret the toxicity resulting from direct contact for most chemicals. Instead, more general means of evaluating exposure-response relationships are used. For example, in a bioassay in which the exposure is to soil, a test organism may be exposed via dermal uptake, and ingestion of the contaminated soil. However, only the concentration in the soil is measured and this concentration is used to evaluate the threshold "exposure" associated with effects. Exposures via each route are never quantified or reported, and may not be needed to interpret the results. Due to a fundamental lack of information to differentiate direct contact exposures from other routes in exposures of ecological receptors, and to interpret this specific exposure route, absorption across the integument is not explicitly addressed by this SLERA.

2.4.3 Inhalation

Inhalation is a potentially complete pathway for wildlife by inhaling airborne particulates or volatilized chemicals. Volatile chemicals are not expected to be present in surface soil in meaningful concentrations for risk, so inhalation of vapors in outdoor air is not a complete pathway. Inhalation is generally considered a relatively minor exposure pathway for wildlife relative to ingestion via soils. An evaluation of risk to receptors via the inhalation pathway may be warranted, however, in cases where volatile organic compounds are COIs and pathways of exposure are complete, including the potential for volatilization of chemicals and exposure to burrowing animals in subsurface soils.

2.5 Assessment Endpoints

An assessment endpoint is "an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes" (USEPA 2003). Clearly defined assessment endpoints help structure an ERA to address management decisions.

Clarity in assessment endpoints is essential to their role in refining the direction of the risk assessment, and in communicating the meaning of the results generated by the SLERA.

USEPA guidance stipulates that assessment endpoints for a SLERA reflect a conservative evaluation of risk, and address any adverse effect potentially resulting from complete exposure pathways linking contaminants to receptors (USEPA 1998). Consistent with USEPA guidance for the SLERA, assessment endpoints are the populations of chosen receptors as inferred from measures related to survival, growth, and reproduction of individuals (USEPA 1998). A summary of assessment endpoints is presented in Table E-3.

The SLERA does not specify the extent or severity of effects of exposure to chemicals on the assessment endpoints for each receptor. Instead, the SLERA identifies those chemicals that have no potential effect on ecological receptors. By using a conservative evaluation of exposure and toxicity, the SLERA identifies those chemicals that require additional evaluation in the BERA, when more realistic and site-specific exposure and toxicity information is considered.

2.6 Preliminary Conceptual Site Model

A CSM is a summary of the sources of contaminants, the physical-chemical processes that control chemical transport and fate over time and space. The PSCR (Integral and Anchor QEA 2012) describes the most current CSM for the south impoundment (Figure E-3) and supporting rationale. It also presents the exposure pathways that potentially lead to exposures of each general category of ecological receptors to COIs. For ecological receptors using the area south of I-10, contact with, inhalation of, and ingestion of contaminated soil within the boundary of the impoundment itself, and in other areas to which COIs may have been transported, creates the potential for exposure (Figure E-4).

3 SCREENING LEVEL EVALUATION AND IDENTIFICATION OF $COPC_E$S

According to USEPA (1997) guidance for conducting SLERAs, only one of the following three conclusions can result from a SLERA for each $COPC_E$:

- There is sufficient information to conclude that ecological risks are negligible
- There is not enough information to make a decision, and additional study may be warranted
- There is adequate information to indicate that a potential for adverse effects exists, and a more thorough assessment is warranted.

A SLERA necessarily applies conservative judgments where there are data gaps or other uncertainties. A conservative approach is used so that a conclusion that ecological risks are negligible can be made with a high degree of confidence.

The screening level exposure estimate and risk calculation is Step 2 of the screening process as defined by USEPA guidance (Figure E-1). Step 2 identifies those Site-related COIs for which there is not enough information to make a decision, or which need additional assessment, and those COIs that represent negligible or no ecological risks. The methods and results used to perform the screening analysis for the data collected to date are presented below. These methods and results have already been presented in Attachment 1 to Soil SAP Addendum 3 (Integral 2011a), and directly reflect the DQOs for the Phase I soil investigation for the south impoundment (Integral 2011b). This document repeats those results, and builds on them to identify $COPC_E$s for the south impoundment.

According to the DQOs for the soil investigation south of I-10 and the analysis path described by Integral (2011b), the analytical approach for the Phase I results of the soil investigation includes the following steps:

- Evaluation of detection frequency. Chemicals detected at a frequency of 5 percent or less are not addressed by subsequent analyses.
- Risk-based screening, consisting of comparison of COI concentrations in surface and shallow subsurface soils to screening levels protective of ecological receptors.

This section presents the results of these analysis steps, and presents the rationale for selection of those COPCEs to be evaluated in the BERA. For the screening comparison, COI concentrations only in surface soils (0 to 6 inches) and shallow subsurface soils (6 to 12 inches) are compared to ecological risk-based screening levels. Risks associated with COPCEs will be evaluated in the BERA for the area south of I-10 according to receptor-specific exposure assumptions, which may include consideration of deeper soils.

3.1 Detection Frequency

COIs with a detection frequency at or below 5 percent of all samples collected in Phase I will not be considered further for the south impoundment area. Detection frequencies for each COI are reported in Table E-4. All of the metals were detected in more than 5 percent of samples, as were dioxin and furan congeners, three of the nine Aroclors, and several semivolatile organic compounds (SVOCs). Those with a detection frequency of 5 percent or less were SVOCs and the two volatile organic compounds on the COI list. The following COIs were detected in 5 percent of samples or less:

- 2,4-Dichlorophenol
- 1,2,3-Trichlorobenzene
- 1,2,4-Trichlorobenzene
- 2,4,5-Trichlorophenol
- 2,4,6-Trichlorophenol
- 2,3,4,6-Tetrachlorophenol
- Pentachlorophenol
- Hexachlorobenzene
- Chloroform.

Although several Aroclors were never detected (Aroclors 1016, 1221, 1232, 1248, 1262, and 1268), any one Aroclor represents a mixture of polychlorinated biphenyls (PCBs), and one or more PCB congeners making up an Aroclor mixture may be present, even if the Aroclor is not detected. Therefore, the detection frequency of zero for these Aroclors does not provide the basis for eliminating any individual PCB congeners from further analysis.

3.2 Ecological Risk-Based Screening Methods

The CSM indicates that terrestrial mammals, reptiles, and birds are the ecological receptor categories of interest for the south impoundment area. To perform a screening evaluation in support of identification of COPCEs, soil screening values protective of birds and mammals were assembled from USEPA's ecological soil screening levels (EcoSSL) (USEPA 2005) (Table E-5). USEPA's EcoSSLs are preferred because they are the result of a rigorous and transparent process involving comprehensive literature assembly and review. Unfortunately, rigorously derived soil screening levels for PCBs and SVOCs were not found (Table E-5).

TCEQ (2006) guidance allows for the use of Texas-specific median background concentrations to screen out COIs when no screening values are available. If concentrations on the Site are below the Texas median background concentration, screening levels may be ignored. If no EcoSSLs or Texas-specific median background concentrations were available, which was the case for all of the SVOCs as well as PCBs, the median value for the Site-specific background concentrations for surface and shallow subsurface soils was used for comparison. These are shown in Tables E-5 and E-6. Consideration of the Site-specific median background concentrations is consistent with TCEQ guidance cited above (TNRCC 2001; TCEQ 2006).

Only screening levels for birds and mammals are used in this document. Soil screening levels for reptiles are generally not available, and, for the purposes of screening, the screening values for birds are considered to be protective of reptiles. In addition, rigorous and technically defensible ecological soil screening levels for dioxins and furans, or toxicity equivalents for dioxins and furans, were not found for this analysis. Therefore, dioxins and furans are not considered in this ecological screen, but are considered COPCEs for the south impoundment area, and will be addressed by the BERA.

Once all of the screening or background median values were compiled, the maximum concentration among all samples from 0 to 6 and 6 to 12 inches were compared to the screening value (Table E-6). Consistent with ERA guidance (USEPA 1998), maximum concentrations or, in the case where the chemical was not detected in all Site samples, one-half the maximum detection limit was used to provide a conservative estimate of exposure concentrations for ecological receptors for the screening evaluation.

3.3 Ecological Risk-Based Screening Results

Results of comparisons of COI concentrations in soils from 0- to 6- and 6- to 12-inch intervals to ecological soil screening values are summarized in Table E-6:

- The following COIs were not found at concentrations greater than the screening value for mammals: aluminum, arsenic, barium, cobalt, manganese, nickel, silver, and vanadium.
- The following COIs were not found at concentrations above the screening value for birds: aluminum, antimony, arsenic, cobalt, manganese, nickel, and silver.

Several metals are present in one or more soil sample from 0 to 6 or 6 to 12 inches at concentrations greater than screening values:

- The following are present at least once at a concentration greater than the screening level for mammals: antimony, cadmium, chromium, copper, mercury, lead, thallium, and zinc.
- The following are present above screening concentrations for birds, in one or more samples: barium, cadmium, chromium, copper, lead, mercury, thallium, vanadium, and zinc.

Screening for antimony, barium, chromium, mercury, and thallium was performed using the Site-specific median background concentrations, because neither ecological risk-based screening values nor Texas median background concentrations were available (Table E-6). From these comparisons, it is evident that magnesium, total PCBs, and all of the SVOCs are present at concentrations greater than the median of the Site-specific background dataset. The exceedances of the median concentration by 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene are not considered likely to be ecologically significant because the maxima for these analytes are nondetects, and the differences from the background medians are very slight (Table E-6).

3.4 Identification of COPCEs for the South Impoundment

To conduct the BERA, it is necessary to identify COPCEs. For the south impoundment, the approach to determining whether each COI is a COPCE is similar to the approach described in Appendix C of the RI/FS Work Plan, and consistent with that approach:

- If the maximum concentration of a COI is greater than the soil screening level or greater than the median background concentration (for those COIs lacking screening levels), the chemical is considered a COPCE if it is bioaccumulative.
- If a COI is not bioaccumulative, then it will not be evaluated in the BERA, even if it exceeds a soil screening level or median background concentration.

All resulting COPCEs will be included in the BERA for the south impoundment.

Potential for bioaccumulation of metals was evaluated using TCEQ guidance, which lists chemicals considered to be bioaccumulative (Table 3-1 in TNRCC [2001] and TCEQ [2006]). Because TCEQ guidance does not address some of the organic COIs, for all of the organic COIs, the log K_{ow} was used as an indicator of bioaccumulation potential. Consistent with USEPA (2008a) guidance, chemicals with log K_{ow}s equal to or greater than 5 were considered to have the potential to bioaccumulate in tissue.

A summary of the decision for each chemical is provided in Table E-7. The following chemicals exceeded a screening concentration or background, but were not selected as COPCEs because they are not considered to be bioaccumulative:

- Antimony. This metal was present at a concentration above the EcoSSL for mammals, but it does not exceed the Texas median background concentration and it is not bioaccumulative, and so it was not considered a COPCE.
- Barium. This metal was present below the EcoSSL for mammals and slightly above the EcoSSL for birds, but is not bioaccumulative, and so was not considered a COPCE. The maximum concentration is below the Site-specific reference envelope value (Integral and Anchor QEA 2012).
- Magnesium. This metal was present above the Site-specific background concentration, but it did not exceed the reference envelope value or the mean in the

Site-specific background soils data. It is not bioaccumulative and is an essential nutrient for many species. It was not considered a COPCE.

- Thallium. This metal was present at concentrations greater than the Texas median background concentration, and greater than Site-specific background, but is not considered bioaccumulative, and so was not considered a COPCE.

A summary of the final COPCEs for the south impoundment is presented in Table E-8.

Because USEPA has requested that additional information be collected to describe COIs in south impoundment soils (Integral 2011a), results of any future sample collection and analyses will be considered before a final determination of COPCEs for the south impoundment is made.

4 SCIENTIFIC MANAGEMENT DECISION POINT

According to USEPA (1997) guidance, the end of Step 2 of the ERA process is an SMDP, and a decision is made about those chemicals for which more information may be needed, and those chemicals for which there is enough information to make a determination of negligible risk. This SLERA concludes that there is not enough information to make a determination about ecological risk for the following chemicals:

- Dioxins and furans
- PCBs
- Bis(2-ethylhexyl)phthalate
- Cadmium
- Chromium
- Copper
- Lead
- Mercury
- Zinc.

For all other chemicals, the available information indicates that ecological risks are negligible.

In this SLERA, uncertainties were mitigated by the use of the following specific conservative approaches, methods, or assumptions:

- Development of a comprehensive COI list for the starting point for screening, as described in Appendix C to the RI/FS Work Plan (Anchor QEA and Integral 2010). On the basis of a list of the priority pollutants possibly in the source material at the Site, a conservative set of criteria was used to identify and define the COIs before the risk-based screening process was applied.
- Use of chemistry information for soils collected at the most likely location of the former south impoundment, which could reasonably be expected to have the highest concentrations of COIs in surface soils, to represent the screening level exposures.
- Use of the maximum concentration of each chemical in soil from within the south impoundment perimeter and within areas accessible to wildlife (surface soils) in the screening.

Results of the Phase I and any subsequent soil investigations will be incorporated into the BERA for the south impoundment to reduce uncertainties and establish a more realistic assessment of ecological risks.

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TABLES

Table E-1
Chemicals of Interest

Class	Chemical
Metals	
	Aluminum
	Antimony
	Arsenic
	Barium
	Cadmium
	Chromium
	Cobalt
	Copper
	Lead
	Magnesium
	Manganese
	Mercury
	Nickel
	Silver
	Thallium
	Vanadium
	Zinc
Dioxins and Furans	
	Dioxins and Furans
Polychlorinated Biphenyls	
	Polychlorinated Biphenyls
Semivolatile Organic Compounds	
	2,4-Dichlorophenol
	2,4,5-Trichlorophenol
	2,4,6-Trichlorophenol
	2,3,4,6-Tetrachlorophenol
	Acenaphthene
	Bis(2-ethylhexyl)phthalate
	Carbazole
	Fluorene
	Hexachlorobenzene
	Naphthalene
	Pentachlorophenol
	Phenanthrene
	Phenol
Volatile Organic Compounds	
	1,2-Dichlorobenzene
	1,3-Dichlorobenzene
	1,4-Dichlorobenzene
	1,2,3-Trichlorobenzene
	1,2,4-Trichlorobenzene
	Chloroform

Table E-2
Ecological Receptor Surrogates for the South Impoundment

Receptor Group	Receptor Surrogate	Feeding Guild	Potentially Present	Representative of One or More Feeding Guilds	High Site Fidelity/Residential	Sensitive or Potentially Highly Exposed	Life History Information Is Readily Available	Additional Considerations
Reptiles								
	Common garter snake	Omnivore (terrestrial)	X	X	X	X	X	
Birds								
	Killdeer	Invertivore (terrestrial)	X	X	X	X	X	Feeds on invertebrate fauna closely associated with soils
Mammals								
	Virginia opossum	Omnivore (terrestrial)	X	X	X	X	X	Terrestrial omnivore with habitat preferences consistent with site conditions
	Pocket gopher	Herbivore	X	X	X	X	X	Burrowing mammal, susceptible to exposure to chemicals of interest in soil

Table E-3
Summary of Receptor Surrogates, Assessment Endpoints, and Risk Questions

Receptor Class	Assessment Endpoint	Risk Questions
Reptiles	Stable or increasing populations of omnivorous reptiles	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth and reproduction of reptiles?
Birds	Stable or increasing populations of terrestrial invertivorous birds	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth, and reproduction of birds? Is the estimated concentration of dioxins and furans, expressed as TEQs, in bird eggs greater than threshold concentrations for reproductive effects in birds?
Mammals	Stable or increasing populations of omnivorous mammals	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth and reproduction of mammals?
	Stable or increasing populations of burrowing mammals	Is exposure via inhalation of volatile COPCs greater than doses known to cause effects on the survival, growth and reproduction of mammals?

Notes
COPC = chemical of potential concern
TEQ = toxic equivalent

Table E-4

Detection Frequencies for Chemicals of Interest

Chemical of Interest	Detection Frequency	
Metals		
Aluminum	72/72	100%
Antimony	58/72	81%
Arsenic	72/72	100%
Barium	72/72	100%
Cadmium	69/72	96%
Chromium	72/72	100%
Cobalt	72/72	100%
Copper	72/72	100%
Lead	72/72	100%
Magnesium	71/71	100%
Manganese	72/72	100%
Mercury	71/72	99%
Nickel	72/72	100%
Silver	10/72	14%
Thallium	42/72	58%
Vanadium	72/72	100%
Zinc	72/72	100%
Dioxins and Furans		
TEQ_{DF}	94/94	100%
Polychlorinated Biphenyls		
Aroclor 1016	0/72	**0%**
Aroclor 1221	0/72	**0%**
Aroclor 1232	0/72	**0%**
Aroclor 1242	7/72	10%
Aroclor 1248	0/72	**0%**
Aroclor 1254	22/72	31%
Aroclor 1260	28/72	39%
Aroclor 1262	0/72	**0%**
Aroclor 1268	0/72	**0%**
Semivolatile Organic Compounds		
2,4-Dichlorophenol	0/72	**0%**
2,4,5-Trichlorophenol	0/72	**0%**
2,4,6-Trichlorophenol	0/72	**0%**
2,3,4,6-Tetrachlorophenol	0/72	**0%**
Acenaphthene	50/70	71%
Bis(2-ethylhexyl)phthalate	51/70	73%
Carbazole	37/70	53%
Fluorene	50/70	71%
Hexachlorobenzene	3/70	**4%**
Naphthalene	36/70	51%
Pentachlorophenol	0/72	**0%**

Table E-4

Detection Frequencies for Chemicals of Interest

Chemical of Interest	Detection Frequency	
Phenanthrene	65/70	93%
Phenol	13/72	18%
Volatile Organic Compounds		
1,2-Dichlorobenzene	14/72	19%
1,3-Dichlorobenzene	15/72	21%
1,4-Dichlorobenzene	13/72	18%
1,2,3-Trichlorobenzene	0/72	**0%**
1,2,4-Trichlorobenzene	2/72	**3%**
Chloroform	2/72	**3%**

Notes

TEQ_{DF} = Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors

Bold typeface indicates that the frequency of detection is less than or equal to 5 percent

Table E-5
Soil Screening Levels Used for Ecological Screening

Analyte	Mammalian Eco-SSL[a]	Avian Eco-SSL[a]	Background Soil Concentration[b]
Metals (mg/kg - dw)			
Aluminum	NA	NA	**30,000**
Antimony	**0.27**	NA	**1**
Arsenic	**46**	**43**	5.9
Barium	**2,000**	NA	**300**
Cadmium	**0.36**	**0.77**	NA
Chromium (total)	NA	NA	**30**
Cobalt	**230**	**120**	7
Copper	**49**	**28**	15
Lead	**56**	**11**	15
Magnesium	NA	NA	NA
Manganese	**4,000**	**4,300**	300
Mercury	NA	NA	**0.04**
Nickel	**130**	**210**	10
Silver	**14**	**4.2**	NA
Thallium	NA	NA	**0.7**
Vanadium	**280**	**7.8**	50
Zinc	**79**	**46**	30
Polychlorinated Biphenyls (µg/kg-dw)			
Aroclor 1242	NA	NA	NA
Aroclor 1254	NA	NA	NA
Aroclor 1260	NA	NA	NA
Total PCBs	NA	NA	**9.5**[d]
Semivolatile Organic Compounds (µg/kg-dw)			
1,2-Dichlorobenzene	NA	NA	**0.048**[c]
1,3-Dichlorobenzene	NA	NA	**0.06**[c]
1,4-Dichlorobenzene	NA	NA	**0.055**[c]
Acenaphthene	NA	NA	**0.7**
Bis(2-ethylhexyl)phthalate	NA	NA	**5.35**
Carbazole	NA	NA	**0.65**
Fluorene	NA	NA	**0.55**
Naphthalene	NA	NA	**1.15**
Phenanthrene	NA	NA	**2.4**
Phenol	NA	NA	**1.4**[e]

Notes

Bold = value was used in the screening evaluation (Table E-6)

a - Value is the minimum value available from the two of the feeding guilds within the taxon evaluated by USEPA (2011).

b - Values for metals are from Texas-Specific Median Background Concentration (Figure 30 TAC §350.51(m)); values for organics are site-specific median background concentration

c - Analyte was never detected in 0- to 12-inch background soils; value shown is the median of the estimated values (i.e., one-half of detection limit) for the chemical in background samples from 0 to 6 inches.

d - This value is the detection limit for individual Aroclors.

e - Detected in 1 of 40 samples.

Table E-6
Ecological Screening Results for Surface and Shallow Subsurface Soils

Chemical of Interest	Maximum Detected Concentration, Surface and Shallow Subsurface Soils (0 to 6 and 6 to 12 inch)	Ecological Screening Value, Mammals [a]	Maximum Exceeds Screening Value	Ecological Screening Value, Birds [a]	Maximum Exceeds Screening Value	Median for Background Soils (0 to 12 inch)	Maximum Exceeds Site-Specific Median Background
Metals (mg/kg - dw)							
Aluminum	11,700	30,000[b]		30,000[b]			
Antimony	1.00 J	0.27	X	1[b]			
Arsenic	5.28 J	46		43			
Barium	413 J	2,000		300[b]	X		
Cadmium	1.28	0.36	X	0.77	X		
Chromium	70.3 J	30[b]	X	30[b]	X		
Cobalt	22.1	230		120			
Copper	121 J	49	X	28	X		
Lead	117 J	56	X	11	X		
Magnesium	9,150	NA	--	NA	--	942	X
Manganese	2,630 J	4,000		4,300			
Mercury	0.156	0.04[b]	X	0.04[b]	X		
Nickel	85.1	130		210			
Silver	0.800 J	14		4.2			
Thallium	9.80 J	0.7[b]	X	0.7[b]	X		
Vanadium	52.1	280		7.8	X		
Zinc	4,160 J	79	X	46	X		
Polychlorinated Biphenyls (µg/kg-dw)							
Total PCBs	427	NA	--	NA	--	9.5	X
Semivolatile Organic Compounds (µg/kg-dw)							
1,2-Dichlorobenzene	0.055 U	NA	--	NA	--	0.048[c]	X
1,3-Dichlorobenzene	0.07 U	NA	--	NA	--	0.06[c]	X
1,4-Dichlorobenzene	0.06 U	NA	--	NA	--	0.055[c]	X
Acenaphthene	88	NA	--	NA	--	0.7	X
Bis(2-ethylhexyl)phthalate	2,200	NA	--	NA	--	5.35	X
Carbazole	48	NA	--	NA	--	0.65	X
Fluorene	46	NA	--	NA	--	0.55	X
Naphthalene	50	NA	--	NA	--	1.15	X
Phenanthrene	450	NA	--	NA	--	2.4	X
Phenol	6.5 U	NA	--	NA	--	1.4[d]	X

Notes

-- = uncertain; no screening value is available for this chemical

NA = no screening value available

U = not detected

X = maximum concentration exceeds screening value

a - USEPA's (2005) EcoSSLs were used, and where they were not available, Texas Median Background concentration is shown (Table E-5)

b - The Texas median background concentration is shown.

c - Analyte was never detected in 0- to 12-inch background soils; value shown is the median of the estimated values (i.e., one-half of detection limit) for the chemical in background samples from 0 to 6 inches.

d - Detected in 1 of 40 samples.

Table E-7
Selection of $COPC_E$s for the South Impoundment

Chemical	Log Kow of Chemical (Organics Only) [a]	Is Chemical Potentially Bioaccumulative? [b]	Maximum Exceeds Avian Screening Value or Background	Maximum Exceeds Mammalian Screening Value or Background	Maintain as COPC for South Impoundment Receptors?		Reason for COPC Decision
					Birds and Reptiles	Mammals	
Metals (mg/kg)							
Aluminum	NA	No	No	No	No	No	Not potentially bioaccumulative
Antimony	NA	No	No	Yes	No	No	Not potentially bioaccumulative
Arsenic	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Barium	NA	No	B	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSL for mammals
Cadmium	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Chromium	NA	Yes	B	B	Yes	Yes	Potentially bioaccumulative, exceeds Texas Median Background
Cobalt	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Copper	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Lead	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Magnesium	NA	No	B	B	No	No	Not potentially bioaccumulative
Manganese	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Mercury	NA	Yes	B	B	Yes	Yes	Potentially bioaccumulative, exceeds Texas Median Background
Nickel	NA	Yes	No	No	No	No	Potentially bioaccumulative, but did not exceed mammal or bird EcoSSLs
Silver	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Thallium	NA	No	B	B	No	No	Not potentially bioaccumulative
Vanadium	NA	No	Yes	No	No	No	Not potentially bioaccumulative, did not exceed mammal EcoSSL
Zinc	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Dioxins/Furans (ng/kg)	>5	Yes	NA	NA	Yes	Yes	Potentially bioaccumulative, indicator chemical group
Polychlorinated Biphenyls (µg/kg)	>5	Yes	B	B	Yes	Yes	Potentially bioaccumulative, detected above background
Semivolatile Organic Compounds (µg/kg)							
2,4-Dichlorophenol	3.06	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,4,5-Trichlorophenol	3.69	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,4,6-Trichlorophenol	3.72	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,3,4,6-Tetrachlorophenol	4.45	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
Acenaphthene	3.92	No [c]	B	B	No	No	Not potentially bioaccumulative
Bis(2-ethylhexyl)phthalate	7.6	Yes	B	B	Yes	Yes	Potentially bioaccumulative, present above the Site-specific background median concentration
Carbazole	3.72	No [c]	B	B	No	No	Not potentially bioaccumulative
Fluorene	4.18	No [c]	B	B	No	No	Not potentially bioaccumulative
Hexachlorobenzene	5.73	Yes	NA	NA	No	No	Detected in less than 5% of samples
Naphthalene	3.3	No [c]	B	B	No	No	Not potentially bioaccumulative
Pentachlorophenol	5.12	Yes	NA	NA	No	No	Detected in less than 5% of samples
Phenanthrene	4.57	No [c]	B	B	No	No	Not potentially bioaccumulative
Phenol	1.46	No	B	B	No	No	Not potentially bioaccumulative

Table E-7
Selection of $COPC_E$s for the South Impoundment

Chemical	Log Kow of Chemical (Organics Only) [a]	Is Chemical Potentially Bioaccumulative? [b]	Maximum Exceeds Avian Screening Value or Background	Maximum Exceeds Mammalian Screening Value or Background	Maintain as COPC for South Impoundment Receptors?		Reason for COPC Decision
					Birds and Reptiles	Mammals	
Volatile Organic Compounds (µg/kg)							
1,2-Dichlorobenzene	3.43	No [c]	B	B	No	No	Not potentially bioaccumulative, maximum concentration was non-detect
1,3-Dichlorobenzene	3.53	No [c]	B	B	No	No	Not potentially bioaccumulative, maximum concentration was non-detect
1,4-Dichlorobenzene	3.44	No [c]	B	B	No	No	Not potentially bioaccumulative, max.concentration was non-detect
1,2,3-Trichlorobenzene	4.05	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
1,2,4-Trichlorobenzene	4.02	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
Chloroform	1.97	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative

Notes

B = Maximum concentration exceeds Texas median background concentration or Site-specific median background concentration

$COPC_E$ = chemical of potential concern for south impoundment ecological receptors

NA = not applicable

TCEQ = Texas Commission on Environmental Quality

a - Log Kow: Octanol-water partition coefficient, the ratio of the concentration of a chemical in octanol and water at equilibrium and at a specified temperature. Octanol is an organic solvent that is used as a surrogate for natural organic matter (e.g., lipids). Values obtained from the HSDB (http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB) or Oak Ridge National Laboratory Risk Assessment Information System (http://rais.ornl.gov/cgi-bin/tox/TOX_select?select=chem)

b - Determination of potential for bioaccumulation from soil is based on TCEQ guidance (TCEQ 2006) or, if chemical is not addressed in guidance, log Kow information is used to determine bioaccumulative potential (as indicated in footnote c), with those chemicals having log Kow>5 being considered potentially bioaccumulative (USEPA 2008).

c - Not provided in TCEQ guidance; log Kow used to determine potential for bioaccumulation as described in footnote b.

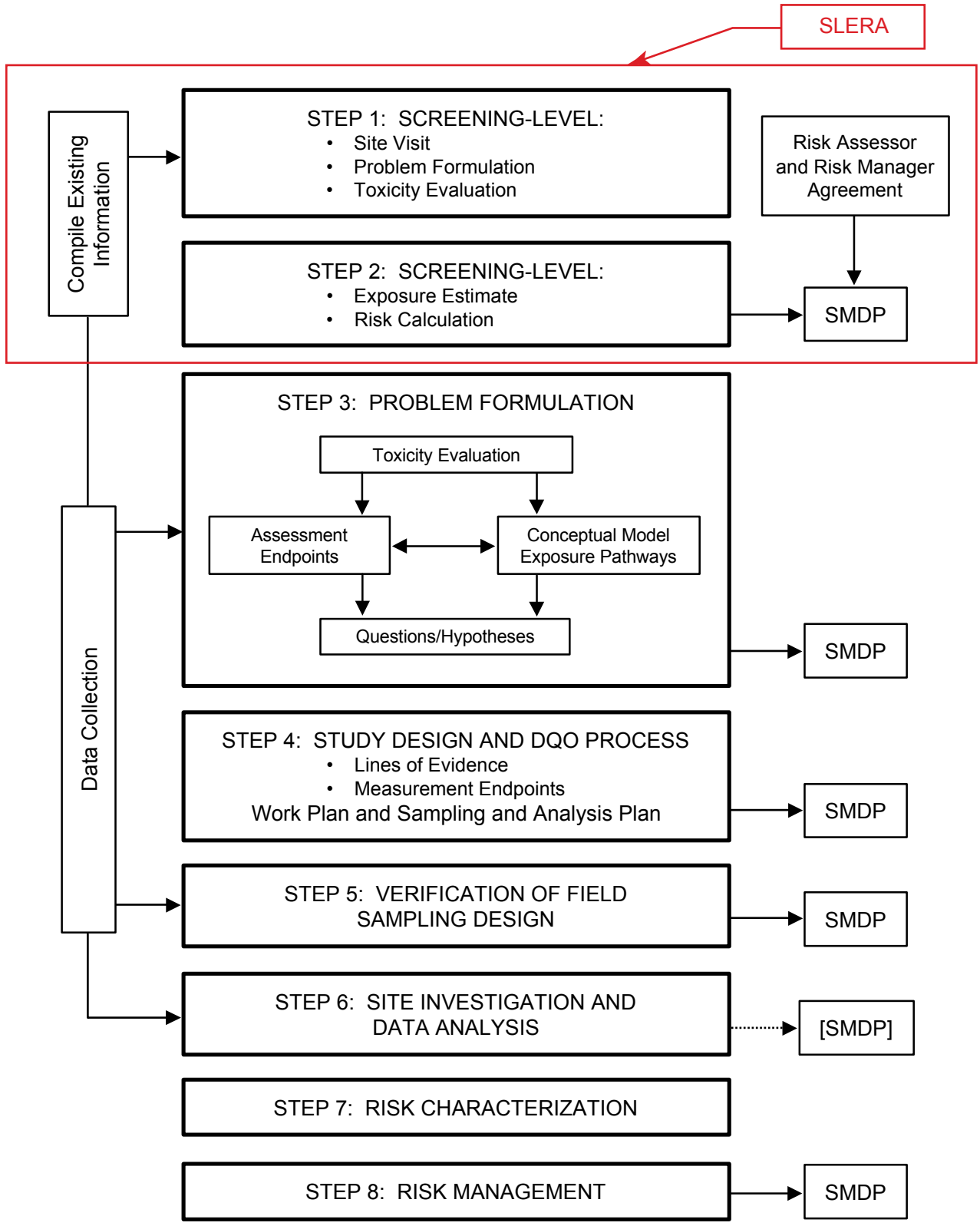
Table E-8
$COPC_E$s for the South Impoundment

Chemical	Reptiles, Birds, and Mammals
Metals	
Cadmium	X
Chromium	X
Copper	X
Lead	X
Mercury	X
Zinc	X
Dioxins/Furans	
Dioxins and Furans	X
Polychlorinated Biphenyls	
Polychlorinated Biphenyls	X
Semivolatile Organic Compounds	
Bis(2-ethylhexyl)phthalate	X

Notes

$COPC_E$ = chemical of potential ecological concern

FIGURES



[SMDP] only if change to the sampling and analysis plan is necessary.

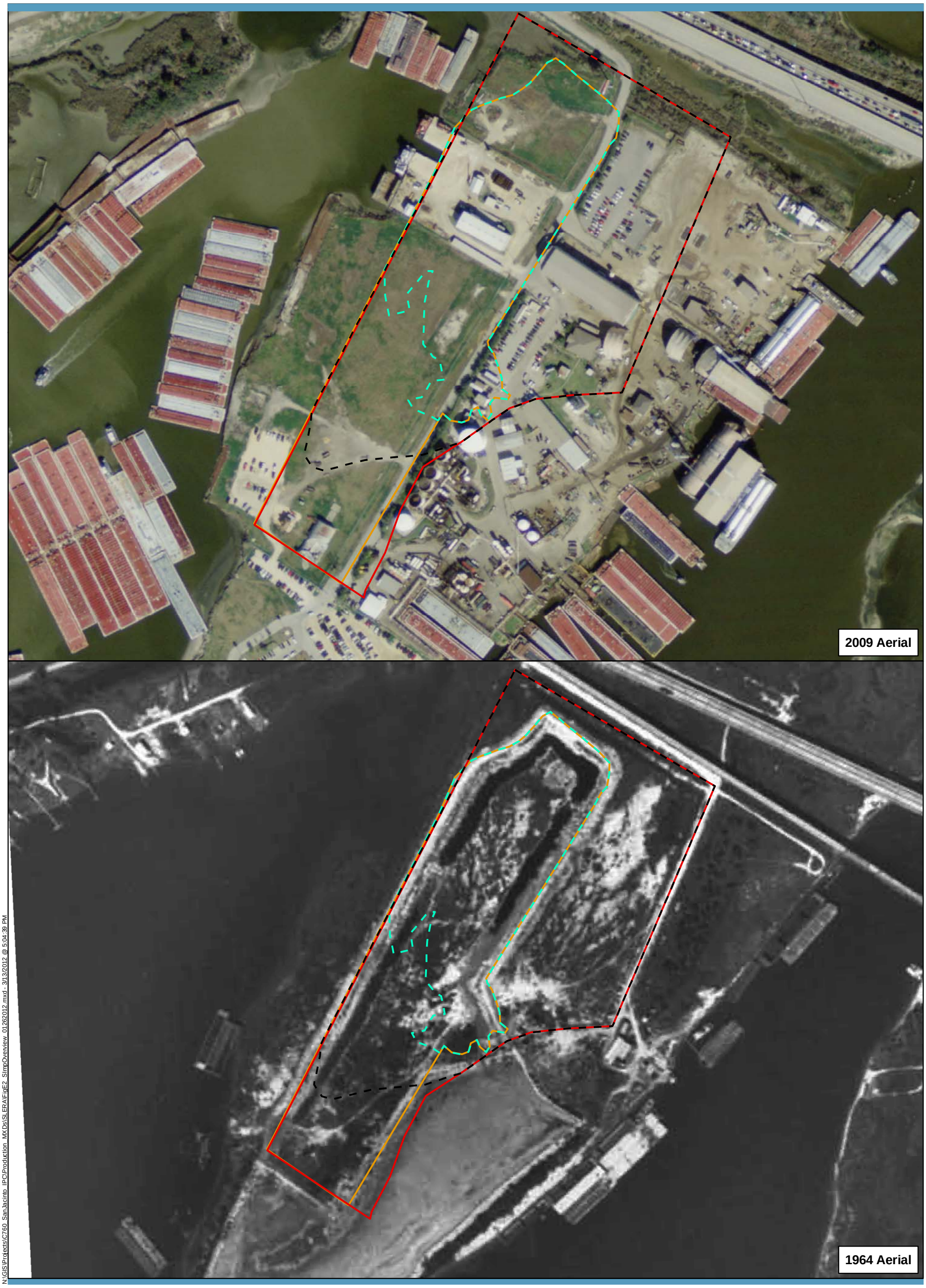
Figure E-1

The Screening Level Ecological Risk Assessment in the Context of the USEPA 8-Step Process for Ecological Risk Assessment

SJRWP SLERA

SJRWP Superfund/IPC

ANCHOR QEA integral consulting inc.



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ANCHOR QEA

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Digitized State Department of Health Hand Drawn Map

Flooded Area on 1966 Aerial Photograph

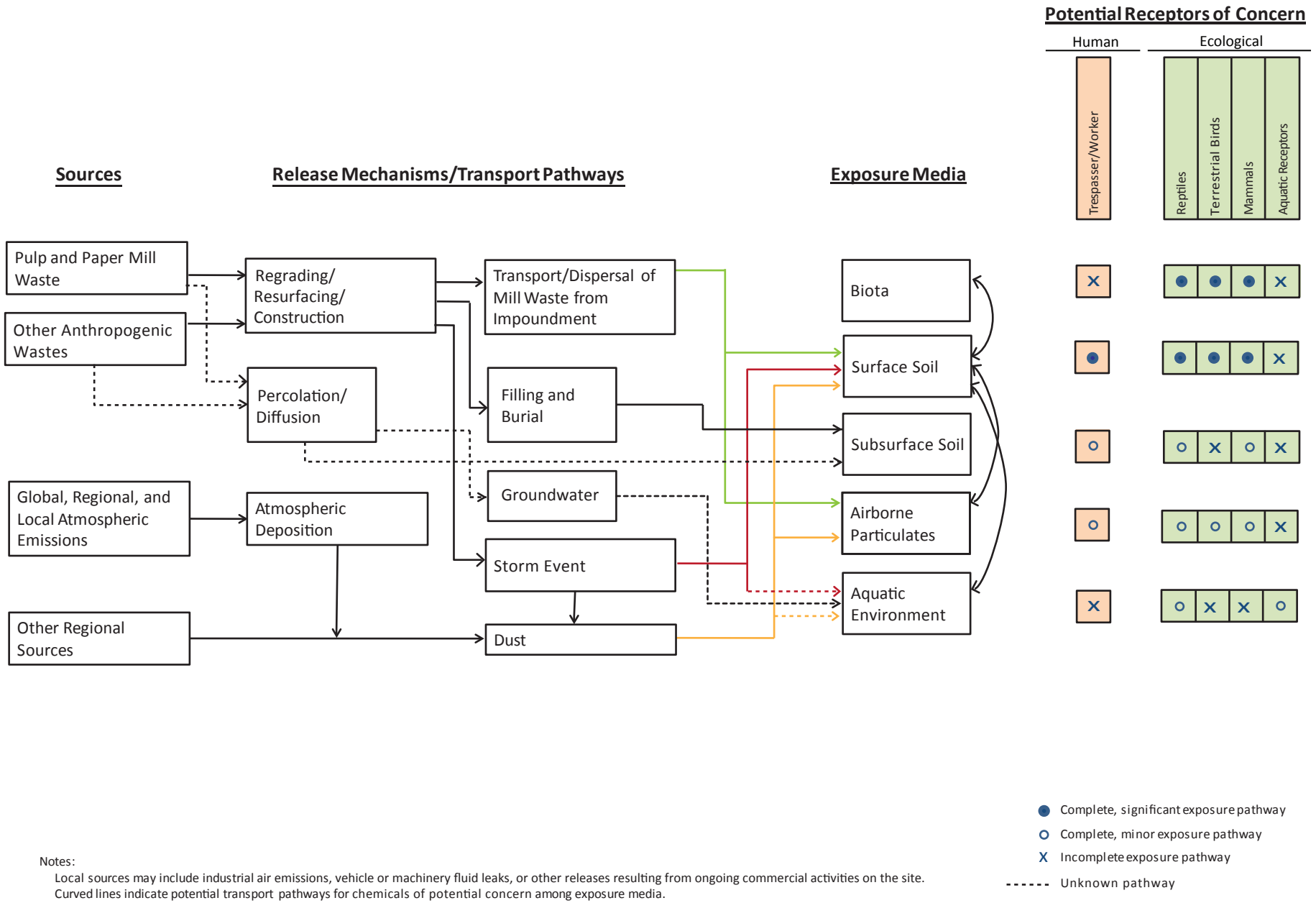
Texas State Department of Health May 1966 Hand-Drawn Map (TSDH 1966) and Soil Investigation Area 4

USGS 1964 Aerial Photograph South Impoundment Perimeter

0 400

Feet

Figure E-2
Soil Investigation Area 4
SJRWP SLERA
SJRWP Superfund/IPC



Notes:

Local sources may include industrial air emissions, vehicle or machinery fluid leaks, or other releases resulting from ongoing commercial activities on the site.

Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.

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Figure E-3

Updated Conceptual Site Model Pathways for the Area South of I-10

SJRWP SLERA

SJRWP Superfund/IPC

Exposure Media	Exposure Routes	Reptiles	Terrestrial Birds[a]	Mammals[a]
			Invertivore	
Biota	Ingestion	●	●	●
Soils	Ingestion	●	●	●
	Direct Contact	○	○	○
	Inhalation	○	○	●[b]

Notes:

● Potentially complete and significant exposure pathway

○ Potentially complete but minor exposure pathway

[a] Mammals and terrestrial birds are assumed not to ingest surface water for drinking, as surface water is estuarine.

[b] Potentially complete and significant for burrowing mammals only (e.g., pocket gopher).

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Figure E-4

Conceptual Site Model for Ecological Exposures:
Exposure Details for Receptor Feeding Guilds and Habitat Associations
SJRWP SLERA
SJRWP Superfund/IPC

APPENDIX F
EPA COMMENTS RELATING TO THE DRAFT BASELINE ECOLOGICAL RISK ASSESSMENT (BERA) DATED MARCH 15, 2012, AND RESPONSES, AND DRAFT-FINAL BERA DATED AUGUST 2012, AND RESPONSES

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	Section	Comment	Response to Comment—Proposed Revision
General Comments			
1		Evaluation of Threatened and Endangered Species: The EPA previously commented (see June 3, 2010 letter regarding review of the draft RI/FS Work Plan and SLERA, Comment 41) that if state or federally listed threatened or endangered wildlife species could occur in the vicinity of the Site, the SERA should designate a surrogate species for the protected species, and base any hazard quotient calculations or risk characterization on the NOAEL TRV (no-observed adverse effect level toxicity reference value) or equivalent. The PRPs agreed with the response and indicated that the text of Appendix B and Attachment Bl would be modified to address the appropriate surrogate species for any listed species that may occur at the Site. Appendix B of the RI/FS work plan generally stated (Section 2.3.2) that the risk assessment for the protected species would not employ the use of surrogates because of the potential to overestimate risk to these listed species, that realistic exposure parameters would be identified for these species, and species specific exposures would be evaluated against the appropriate TRVs in the BERA. The BERA did imply or state (Section 3.4.4) that the sandpiper would make an appropriate representative for the white-faced ibis, a State-threatened species, due to similar feeding/foraging strategies. Because the NOAEL hazard quotients for copper [central tendency (CT) = 2; reasonable maximum exposure concentration (RM) = 3] and TEQ_{DFP} (CT = 10; RM = 30) were greater than 1, the assessment shall include a more robust discussion/ analysis (TEQ_{DFP} denotes the toxicity equivalent (TEQ) concentrations calculated using dioxins and furans and dioxin-like PCBs). The text simply states that the ibis would only be an occasional visitor to the Site and its exposure potential is considered low.	Although endangered species are addressed by the draft Baseline Ecological Risk Assessment (BERA) in the Problem Formulation (Section 3.4.4) and in the Uncertainty Analysis (Section 7.1), exposures were not quantified for those endangered species that could occasionally occur on the site (brown pelican, bald eagle, and white-faced ibis). Because the risk assessment concluded that only those organisms with small home ranges in areas near the northern impoundments are potentially exposed at levels associated with risk, a quantitative evaluation of exposure for these species was not conducted, because they have large home ranges that are much greater than the area of the San Jacinto River Waste Pits site (Site). To address the risk to protected species with greater specificity, text will be added to the problem formulation section, to the exposure assessment, to Section 6 (as Section 6.7), to the discussion in Section 7, and to Appendix A. Revisions will more clearly address exposure and risk as a function of home range size relative to that of the receptor surrogates.
2		Post TCRA (Time-Critical Removal Action) –Scenarios: Hazard quotient calculations were presented for the baseline site (before placement of the TCRA), and after TCRA placement. For the post-TCRA analysis, the evaluation assumed that $COPC_E$ (chemical of potential ecological concern) concentrations in sediments within the TCRA footprint (i.e., sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of 1-10) are equal to the median concentration of the chemical in the upstream background sediment dataset or the background soil dataset. Additionally, pre-TCRA tissue concentrations were used in post-TCRA analyses. The following shall be considered: the presumption that the Site post-TCRA will continue to remain devoid of habitat assumes that the Site will be maintained to prevent this from happening. The assessment shall consider that the Site post-TCRA will develop habitat over time.	The text in Section 3.4.3 that describes the post-TCRA habitat is not intended to suggest an assumption in the BERA that the post-TCRA environment will not provide habitat. The post-TCRA exposure scenarios assume that species will use the capped area as they have under baseline conditions. The only assumption that differs is the concentrations of $COPC_E$s in sediments within the original 1966 impoundment perimeter. Text will be edited in Section 3.4.3 and added to Section 3.8.4.3 to clarify this.
3		Estimating surface water concentrations of COPCs from sediments shall be considered a major data gap and point of uncertainty, and clarified as such in the report.	In a meeting with the U.S. Environmental Protection Agency (USEPA) and other agencies on July 18, 2012, it was discussed and agreed that this comment was not intended to indicate a requirement for additional sampling, but that estimation of chemicals of potential concern (COPCs) from sediments constitutes an uncertainty that should be further discussed. Discussion will be added to Section 7 to better describe uncertainty associated with the methods used to estimate water concentrations.
4		Figures depicting tissue sample locations shall include points at which the actual samples used in the analyses were located. The reader is unable to determine the spatial relationship between individual samples as currently depicted.	Figures 5, 6, 9, and 10 in the Field Sampling Report: Tissue Study (Integral 2011) depict the actual locations of crab and large fish tissue samples. These maps will be included in Section 4.2.5 of the draft final BERA. Small fish and clams were collected on transects; current maps depict the most specific representation of collection area available.

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	Section	Comment	Response to Comment—Proposed Revision
5		The presentation of the results of the BERA made it difficult to independently evaluate the risk conclusions. In particular, it would have been useful for the results to be presented in tables that included the site specific data along with the TRV or baseline values used for the assessment. By presenting the data in different locations, and by presenting primarily summaries rather than the raw data and calculations used to generate the summary data, it was challenging to trace the conclusions made in the BERA. A revision to this document shall include summary tables sufficient to allow reviewers to follow the assumptions made in the BERA.	At a meeting on July 18, 2012, USEPA clarified that this comment is intended to capture the overall sense conveyed by requests in detailed comments that in some areas, the reader would benefit from additional illustration of the methods. The reviewers suggested that the response include building from Table 3-10, which is a summary of assessment endpoints, to provide more of a "road map" to how the analyses were performed. Reviewers are reminded that Table 3-11 builds from Table 3-10 to list the lines of evidence, measures of exposure and measures of effect to be used to address each assessment endpoint. To provide additional illustration of methods, clarifications will be made to tables and figures including those called for in comments 17, 18, 26, 34, 35, 54, 65, and 72. Additional information in the form of exhibits will be added to show examples of how calculations were performed for the wildlife exposure model (to address this comment) and the bird-egg exposure model (to address comment 39 and 42).
6		It is not clear what criteria were used in the selection of toxicity references used to develop the TRVs for benthic invertebrates. References should have been prioritized by endpoint, life stage of receptor, habitat of receptor, and duration of test. Some of the references may not be appropriate for derivation of the TRV for this site (particularly those based on freshwater, acute tests). The report shall provide the selection criteria for the reference studies used.	The reader is referred to Sections 4.1 and 5.3 of the text, which describe the iterative approach to identification of toxicity reference values (TRVs) for benthos, and Section 1.4.1 of Appendix B, where the specific set of considerations used in selecting a TRV for any given chemical, including for those chemicals lacking sediment benchmarks, is discussed. Bulk sediment concentrations addressing community endpoints such as sediment benchmarks were preferred. In fact, TCEQ's preferred screening values were used for those $COPC_E$s for which they were available. If sediment quality guidelines (SQGs) were not available, ambient water quality criteria (AWQC) were used and compared to estimated pore water concentrations. Only when neither SQGs nor AWQC were available were other types of toxicity data used (i.e., from USEPA's ECOTOX database). The most protective value from the available ECOTOX data was selected from those studies addressing marine invertebrates; results for freshwater species were not used. This general approach is described in Section 5.3, and Section 5.3 cites Appendix B for details. Appendix B presents details for selection of toxicity values for each $COPC_E$ for benthic invertebrates. Specific information on the origin and derivation of each value is provided in Table 5-1. The BERA did not use a general literature search for each $COPC_E$, because concentrations of chemicals in sediments or estimated in sediment pore water were generally below the screening-level values used as described above. If concentrations of COPCs are generally below these broadly protective screening values, additional evaluation of the literature is not warranted. For dioxins and furans, the approach to evaluation of toxicity to benthos was described in detail in Attachment B2 of Appendix B to the RI/FS Work Plan; that information is also cited in BERA Section 5.3 and summarized or repeated in Appendix B of the BERA.

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7		The assumption that the exposure of receptors post-TCRA will be at background levels for soil and sediment for areas outside the containment area is questionable. The report shall provide justification for why the sediment outside the footprint of the cap may already be at the upstream concentrations.	The post-TCRA analysis does not assume that exposures outside the containment area will be at concentrations comparable to upstream. The post-TCRA analysis only replaces samples within the TCRA footprint with the median of upstream concentrations. Data outside the footprint is not replaced; the reader is referred to Section 3.8.1: "Using the general assumption that $COPC_E$ concentrations in sediments within the TCRA footprint are equal to the median concentration of the chemical in the upstream background sediment dataset." Please also see Section 3.8.4.3, which states "sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of I-10 are eliminated from the dataset used to estimate EPCs, and replaced with the median concentration of the chemical in the upstream background sediment dataset or from the background soil dataset, as appropriate." In a meeting with USEPA on July 18, 2012, reviewers clarified that their concern was that this assumption may be wrong (i.e., that it is unknown what conditions will evolve in terms of sediment chemistry on the cap), and that this uncertainty should be more clearly stated. Text will be added to Section 3.8.4.3 to highlight that the assumption cannot be verified with existing information. Section 7 will include a related discussion.
8		Statements that surface water quality criteria (a typical ARAR), derived to be protective of human and ecological receptors "should not override site-specific values". It shall be clarified whether or not this statement implies that site-specific values are equal to or more conservative than any ARARs. If not, these statements shall be deleted considering the requirements for ARARs and that the site is located in a dynamic and complex environment, where adequate site-specific exposure and risk assessment is difficult, at best.	The language in quotations in the comment could not be found anywhere in the document or appendices. In the meeting with USEPA on July 18, 2012, reviewers clarified that this comment was not intended for the BERA, and does not need to be specifically addressed in the context of this document.
9		The report shall include the rationale for the assumptions and conclusions included in the BERA so that they are transparent and understandable, and conservatism is demonstrated.	In the meeting with USEPA on July 18, 2012, reviewers reiterated that there was a general sense that the document should detail and provide more discussion of assumptions. Reviewers are referred to Tables 3-12 and 4-2 for key exposure assumptions; Section 4 for a narrative description of all exposure assessment methods and assumptions; and Appendix B for underlying information supporting the toxicity evaluation.
10		The report shall provide/expand its description and evaluation of food chain implications in the BERA.	It's unclear whether the comment refers to issues related to bioaccumulation of chemicals, or to changes in energy transfer across the aquatic community that could result from risks to lower trophic levels. Regardless, data for the Site do not include information to support a description or further discussion of the food web. The study and description of aquatic food webs require certain specific types of data, such as analysis of fish stomach contents or stable isotopes of nitrogen in a variety of organisms from a specified area and time. Data of this type have not been collected for the remedial investigation for this Site. It is therefore not appropriate to expand on "food chain implications" in the BERA. A section will be added to the document (as Section 6.7) to address this comment. The Technical Memorandum on Bioaccumulation Modeling (Integral 2010a) provides an extensive discussion of dioxin and furan bioaccumulation in fish, aquatic invertebrates and birds based on the published literature and analysis of data for the Houston Ship Channel. Summary information will be added as Section 6.8. The reviewer is referred to Integral (2010a) for details.

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11	List of Acronyms	A definition for reasonable maximum exposure (RME) shall be added to the acronym list.	Where the acronym RME is used, it will be will be changed to RM (reasonable maximum, for consistency with "CT" for central tendency.
12	2.1	Site Setting and General Conceptual Site Models: The report states that other sources of dioxins and furans are present on the site. The report shall describe these sources.	Additional information will be included in Section 2.1 about other chemical sources on the Site. However, this topic is to be addressed in greater detail in the remedial investigation report, as required by USEPA (1988).
13	3.4	Ecosystems Potentially at Risk: Protections under the Bald and Golden Eagle Protection Act are similar in nature to that of the Endangered Species Act. As such, any surrogate (for Bald Eagle) risk characterization shall be done by comparing exposure to the NOAEL, rather than the LOAEL as presented here in the text.	Please see the response to comment 1. Risk characterization for protected species will be addressed in Section 6.
14	3.3.4	Endangered and Threatened Species at the Site: The report notes that the alligator snapping turtle is on the state list. The alligator snapping turtle's life history and occurrence shall be discussed as the other listed species are in the following paragraphs.	Text will be added to Section 3.4.4 to address the life history and occurrence of the alligator snapping turtle.
15	3.8.4.1	Calculation of Hazard Quotients: Disagree with the assertion that exposures resulting in HQ_L<1 should be characterized as "negligible." Chronic exposure in the site setting to concentrations between the NOAEL and LOAEL could result in some risk. Acceptable and "negligible" risk characterizations shall be limited to those with HQ_N<1. Also, while not being quantified, risks of mixtures of COPCs shall be addressed in the uncertainty section of this document.	The following proposal was discussed with USEPA and other reviewers on July 18, 2012: The language in this section will be changed to indicate the following interpretation: • HQ_N < 1 = risk is negligible • HQ_N > 1 and HQ_L < 1 = risk is very low • HQ_L > 1 = risk is considered present, and additional evaluation is needed to address the assessment endpoint The concept of the assessment endpoint will be specifically addressed by the additional language supporting interpretation of the HQ. In general, assessment endpoints are populations or communities of organisms, while the basis for the HQ is nearly always an individual-level TRV. Therefore, it is not appropriate to conclude or imply that an HQ_N > 1 signifies risk. To better describe risk in the situation where $HQ_N > 1 > HQ_L$, additional context will be provided with discussion of the toxicity information that is the basis for the TRV. In addition, statements about risk for individual COPC–receptor pairs will make the distinction between the individual and the population. The revised approach will be presented in Sections 3.8 and 6.1. Language will be added to Section 7 to address uncertainties regarding the consequences of exposure to chemical mixtures.
16	3.8.4.5	Comparison of Site Risks to Background: The BERA refers to upstream background in a dynamic, tidal setting (Table 6-2, 6-7, 6-8); but no description of the samples that constitute background levels is provided. The report shall provide this description.	A more detailed description of the background data set will be provided in Section 2.2.1.
17	4.1.1	Estimated Water Concentrations (Exposure of Benthic Macro-invertebrates): It appears that in Equation 4-2, the f_{oc} used is sample-specific. The report shall confirm this. Also, as this section deals with estimation of pore water concentrations, it shall be titled as such.	Clarification will be provided as requested.

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18	4.1.3	Results of the Benthic Macro-invertebrate Exposure Evaluation: The BERA shall provide a table that summarizes the estimated sediment pore water concentrations (i.e., mean, maximum, and minimum number of samples) for the various $COPC_E$s evaluated in this manner for the benthic exposure pathway.	A table will be added to Section 4.1.3 to provide estimated pore water concentrations.
19	4.2.1	$COPC_E$ Concentrations in Fish Diets: The referenced citation (Meador et al. 2010) shall reflect a 2011 date.	Clarification will be provided as requested.
20	4.2.2	Estimated Concentrations of Selected $COPC_E$s in Surface Water: Table 4-3 displays the sediment SWAC (surface area-weighted average concentration) and the estimated surface water concentration for a number of $COPC_E$s. The methodology for calculating the values is not necessarily transparent. By way of example, the report shall provide a table that displays the calculations for lead and nickel.	The reviewer is referred to Equation 4-3. A column will be added to Table 4-3 with the partition coefficients in between the SWAC and the surface water concentrations.
21	4.3.1	Wildlife Exposure Model: Looking at the values for sediment (or soil) ingestion for the various wildlife receptors in Table 3-12, we assume that the Fs value is intended to be the fraction of the diet that is soil/ sediment and that the units column should be blank. The report shall clarify/confirm this.	Tracking units of parameters in the wildlife exposure model improves transparency of the calculations. Units express important information about the basis for the value and will not be removed from Table 3-12.
22	4.3.1.2	Relative Bioavailability Adjustment Factor: For the wildlife exposure model, the 2,3,7,8-TCDD concentration was multiplied by a relative bioavailability factor (RBA) based on a study by Nosek et al. (1992). In this study, adult ring-necked pheasant hens were administered a single dose of a suspension of TCDD radio-labeled earthworms, soil, paper mill sludge, or crickets. Radioactivity remaining in the bird carcass after 24-hours was measured. This adjustment applied to $TEQ_{DF\ B}$ for sediment and soil at the shoreline, sediment outside of the western cell, shoreline background, post-TCRA shoreline, and soils north of IH-10. For tissue, this adjustment applied to $TEQ_{DF,B}$ for common rangia (site-wide and background) and blue crab (site-wide and background). Additionally, this adjustment applied to $TEQ_{DF,B}$ and $TEQ_{DF,M}$ for terrestrial invertebrates north of IH-10 and the peninsula only. It is unclear that the single exposure and uptake evaluation (after only 24 hr) utilized in the Nosek et al. study sufficiently represents reality (e.g., normal digestive tract residence time). We do not support the use of the referenced RBAs for the following reasons: a. The bioavailability study is not site-specific; b. Uncertainty regarding the dose duration and measurement time (was steady state achieved?); c. Selective uptake of TCDD in bird tissues; and d. Uncertainty in the TCDD dose concentration compared with prey/media concentrations at the San Jacinto River Site.	In the meeting with USEPA on July 18, 2012, it was agreed that additional information from the literature supporting the approach discussed in the comment will be included, and that a specific analysis of the exposure to birds without using the RBAs will be presented in the uncertainty analysis. Text will also be added to Section 4.3.1.2 to describe the uncertainty analysis.

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		The referenced relative bioavailability factor shall not used, and shall be deleted from the report.	
23	4.3.1.3	Unit Conversions: Regarding the conversion of tissue concentrations expressed as wet weight to dry weight, the text shall indicate that this step was already performed (where appropriate) for each tissue sample based on the percent moisture/solids determined by the lab, and that the exposure point concentrations in Appendix C were determined after this conversion.	The reviewer is referred to Section 4.3.1.3, "Before calculating EPCs for tissue on a dry weight basis, wet weight concentrations in individual samples are first converted to dry weight concentrations using the fractional solids data for the same sample if available; if solids data is not available, the average fraction of solids data for the given species is used." Also, please note not all tissue EPCs are in dry weight, there is a mixture and that is explained by footnote a in Appendix C.
24	4.3.1.5.1	Estimating $COPC_E$ Concentrations in Plants (Concentrations of $COPC_E$s in Foods of Alligator Snapping Turtle, Killdeer, Raccoon, and Marsh Rice Rat): The full reference for the Staples et al. (1997) citation was not provided. The report shall provide this reference to the reference section.	Clarification will be provided as requested.
25	4.3.1.5.2	Estimating $COPC_E$: Concentrations in Soil Invertebrates: Soil-to-invertebrate bioaccumulation factors (BAFs) for nickel and thallium were obtained from EPA (1999b) and are provided in Table 4-9. The BAFs are presented on a wet-weight basis in the EPA reference. Because the mammalian dose calculations are performed on a dry-weight basis, it is not clear if the estimated tissue concentrations were converted to dry weight. The report shall clarify this and indicate the assumed moisture content.	Reviewers have correctly identified an error: The tissue concentration resulting from application of bioconcentration factors (BCFs) were not converted to dry weight concentrations as they should have been for nickel and thallium. A correction factor will be applied using the assumed moisture content of 84 percent in earthworms from USEPA (1993). The correction will be made to the analysis and tables will be updated to reflect revised HQs for receptors that are affected by this correction. Thallium is not a COPC for this BERA, but preliminary evaluation of soils in the south impoundment suggested that it could be a $COPC_E$ for that area. Thallium was inadvertently included in the $COPC_E$ list for this analysis. Information for this chemical will be removed from Table 4-9 and the text.
26	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations (based on Site soil concentrations) for mercury. According to the BERA discussion and Table 4-9, an uptake factor of 3.1 was used for soil concentrations less than or equal to 1.5 mg/kg, and an uptake factor of 0.7 was used for soil concentrations greater than 1.5 mg/kg. Because these BAF values were applied to individual surface soil sample locations, the report shall add information in Appendix C that indicates the predicted CT and RM exposure concentrations for mercury for soil invertebrates.	Additional detail for the estimated CT and RM mercury concentrations in invertebrates will be added to the end of Table C-1, as requested.
27	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Regarding PCBs, the discussion indicates congener-specific models were not used to estimate invertebrate concentrations because there are no PCB congener data for soils at the Site. This is confusing because Table 4-12 indicates $TEQ_{P,B}$ values for the killdeer, Table 6-5 indicates hazard quotients for $TEQ_{P,B}$ for the killdeer, Table 6-9 indicates hazard quotients for $TEQ_{P,M}$ for the marsh rice rat and raccoon, and Table C-l indicates $TEQ_{P,B}$ and $TEQ_{P,M}$ values for soils north of IH-10. The report shall clarify and indicate how TEQ_P was evaluated for terrestrial receptors.	Text will be added to provide clarification, as requested. Dioxin-like PCB congeners were analyzed in TxDOT soils and were used to calculate TEQ_P for use in the exposure model. The text in this section is intended to convey that to evaluate exposure to total PCBs, the full suite of PCB congeners in soils was not available in order to build a congener-specific model, so a regression relationship for total PCBs using total PCBs as sum of Aroclors in soils was used as the basis for deriving an estimate of total PCBs in invertebrates. This will be clarified in Section 4.3.1.5.

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28	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Paired soil and earthworm tissue dioxin and furan data (n = 6) from the St. Regis Paper Company Superfund Site in Cass Lake, Minnesota were used to develop a series of regression and correlation relationships for dioxin and furan congeners. These were used to estimate dioxin and furan concentrations in soil invertebrate tissue for use in the wildlife exposure model for the killdeer and raccoon. For this analysis, P-values ≤ 0.1 were considered statistically significant, and significant regression relationships between soil and tissue were developed for 11 of the 17 congeners. For the remaining 6 congeners, correlation relationships were determined with other congeners. The resulting estimated concentrations of dioxins and furans (TEQ_{DF}) in terrestrial invertebrate tissue for the raccoon or killdeer exposure scenario are shown in Table D-6. Although Sample et al. (1996) is mentioned in the discussion, there is relatively little discussion of alternative approaches. Given the small sample size and the higher than normal threshold for the determination of statistical significance, the adequacy of this approach for estimating invertebrate dioxin/furan concentrations is questionable. The report shall compare/contrast this approach generally with other relevant dioxin/furan invertebrate uptake estimates in the peer-reviewed and/or CERCLA specific literature.	Additional information from other sites or publications will be added to Appendix D, as available. Results of the evaluation of additional information will be used to qualitatively address related uncertainties.
29	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: The regression and correlation relationships developed from the Cass Lake Superfund site would not be expected to accurately predict soil invertebrate tissue concentrations at the San Jacinto River Site because the range of dioxin and furan concentrations in the six Cass Lake soil samples is much lower, especially for 2,3,7,8-TCDD and 2,3,7,8-TCDF. Additionally, the ratios between congeners in soils from the Cass Lake site are very different from congener ratios at the San Jacinto River Site. For the Cass Lake site, the highest 2,3,7,8-TCDD concentration was 1.83 ng/kg, and the highest 2,3,7,8-TCDF concentration was 11.3 ng/kg (Table *D-1*). In contrast, at the San Jacinto River Site, the highest soil 2,3,7,8-TCDD concentration was 8,650 ng/kg, and the highest 2,3,7,8-TCDF concentration was 20,600 ng/ kg (Table 6-17 in the Preliminary Site Characterization Report). According to Appendix D, the 2,3,7,8-TCDD congener was not detected in 5/6 of the Cass Lake earthworm samples. In the one sample where 2,3,7,8-TCDD was detected in tissue, it was not detected in soil. Because no statistically significant relationship between soil and earthworm concentrations was identified for some congeners, a correlation approach was used, which compared the ratio of congener concentrations in earthworm tissue. The ratio between concentrations of 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD was used to predict the 2,3.7,8-TCDF concentration in invertebrate tissue. For the Cass Lake site, the average 1,2,3,6,7,8-HxCDD concentration in soil was about 50 times greater than the concentration of 2,3,7,8-TCDF in soil. In contrast at the San Jacinto River Site, the average TCDF concentration in Area 3 soils was over 3,200 times the average 1,2,3,6,7,8-HxCDD concentration in soils (Table 6-17 in PSCR). This suggests that the use of the Cass Lake soil data will greatly underestimate the concentration of TCDF in invertebrate tissue at the San Jacinto River Site. Given the significant difference in soil concentrations for TCDD and TCDF, and the uncertainty associated with the ratio approach, the adequacy of this approach for	The overall range of concentrations in the “soil” dataset for the Site is highly skewed by the samples collected from within the original 1966 perimeter of the impoundments north of I-10. Similarly, the dioxin and furan fingerprints of soils within this area, which are characterized by large fractions of the tetrachlorinated congeners, are substantially different from those of soil samples collected outside the impoundments. The concentration ranges of 2,3,7,8-TCDD and 2,3,7,8-TCDF are greater in the impoundments north of I-10 than in Cass Lake soil. However, the ranges and central tendencies of concentrations of these congeners outside of the impoundments are quite similar to those of 2,3,7,8-TCDD and 2,3,7,8-TCDF in Cass Lake soils. This will be clarified in Appendix D. The ratios of maximum values of TCDF to TCDD in Cass Lake soil is 6; in the San Jacinto data, this ratio is approximately 3 for soils both inside and outside of the impoundments. Ratios of congeners for both the Cass Lake and SJRWP data sets inside and outside of the impoundments will be more clearly described in Appendix D. 2,3,7,8-TCDD was not detected in four of six earthworm tissue samples (U-qualified) and was estimated in one sample (J-qualified). In Cass Lake soil, 2,3,7,8-TCDD was not detected in three samples, and was estimated in two. Nevertheless, a significant regression relationship was derived for this congener. Uncertainties associated with the censored data for 2,3,7,8-TCDD will be discussed in revisions to Appendix D. We agree with the comment that 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD ratios for soils within the impoundment are high relative to these ratios in the Cass Lake data set. Therefore, significant correlations with another congener having concentrations more similar to TCDF and with a significant regression relationship will be applied to predict TCDF concentrations in soils within the 1966 impoundment perimeter. Revisions will be made to Appendix D to describe these changes in the analytical approach and results. Consistent with response to comment 28, additional discussion will be provided regarding the appropriateness of this approach.

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		estimating invertebrate dioxin/furan concentrations is questionable. The report shall compare/contrast this approach generally with other relevant dioxin/furan invertebrate uptake estimates in the peer-reviewed and/or CERCLA specific literature.	
30	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: There is a statement in Section 2.1 of Appendix D that "the ranges of dioxin and furan concentrations in soil at the Cass Lake site were similar to the range of concentrations in soils at the San Jacinto River site." This shall be revised. The total TEQ ranges may be similar, but the individual congener ranges were not.	Clarification will be provided as requested; please see response to Comment 29 indicating that a more specific analysis of soils inside and outside of the impoundment will be added to discuss relationships with the Cass Lake soil data set.
31	4.3.1.6	Wildlife Exposure Units: Figure 4-9 depicts the exposure areas and samples used for the killdeer evaluation. The report shall explain why all of the area on the west side of the upland sand separation area was used for the assessment when surface soil data was not available for the far western third of the property. Additionally, the report shall state whether this inclusion was conservative.	Exposure areas were determined by considering the areas that constitute appropriate and accessible habitats. The exposure area selected reflects all available data, because the area use factor (AUF) is set to 1 for this receptor, per Table 4-11, which is the most conservative possible approach. Clarification will be provided as requested.
32	4.3.1.6	Wildlife Exposure Units: Figure 4-10 depicts the exposure areas and samples used for the raccoon evaluation. Very limited soil/sediment data was available for these areas, and clams and small fish were not collected in this area. The report shall explain why all of the area along the west shoreline of the Southern Impoundment and along the eastern shoreline on the land mass across the Old River Channel (and south of IH-10) was used for the assessment. Additionally, the report shall state whether this inclusion was conservative and how will it be integrated with an ecological assessment for the Southern Impoundment.	The raccoon is not a receptor for the south impoundment; the SLERA for the south impoundment is presented in Appendix E. To address risks to raccoon for the northern impoundments and surrounding aquatic environment, shoreline sediments and the tissue samples of aquatic biota within the exposure area for the impoundments north of I-10 and surrounding aquatic habitats were included. In preparation of the sampling and analysis plans, the sediment and tissue sampling designs explicitly considered the risk assessments, and these considerations are described in the data quality objectives provided in Section 1 of each of those two sampling and analysis plans (SAPs). Based on the results of surface sediment samples collected in the Old River to date, concentrations in beach sediments there would be expected to be low. Therefore, the sediment and tissue samples used in the BERA, as a whole have a conservative bias, because they are focused in areas adjacent to the source material that was in the aquatic environment north of I-10 at the time of sampling.
33	4.3.1.6	Wildlife Exposure Units: Similarly, Figure 4-11 depicts the exposure areas and samples used for the great blue heron, spotted sandpiper, and marsh rice rat evaluations. Very limited sediment data was available for the areas south of IH-10, and clams and small fish were only collected in an area along the east side of the river channel shoreline (and south of the IH-10 bridge). It is not clear how data from these areas will be incorporated into the exposure calculations. The report shall clarify this. Additionally, the report shall state whether this inclusion was conservative and how will it be integrated with an ecological assessment for the Southern impoundment.	The exposure area selected reflects all available data, because the AUF is set to 1 for these receptors, per Table 4-11. This is the most conservative possible approach. Clarification will be provided as requested. The reviewer is referred to the second paragraph of the Introduction (Section 1), which addresses the ecological risk assessment process for the south impoundment area. This process begins with the SLERA, which is presented in Appendix E (to be modified according to comments 74 and 75). The approach was developed consistent with the conceptual site models (CSMs) for the site, updated in the Preliminary Site Characterization Report (PSCR).

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34	4.3.1.7	Calculation of Exposure Point Concentrations: Appendix C shall be amended to include the surface water CT and RM exposure point concentrations for TEQs and Total PCBs that were used for determining the bird dose (i.e., surface water ingestion).	Details for $TEQ_{DF,B}$ will be provided in Appendix C as requested. Because PCBs are bioaccumulative, animals receive the majority of ingestion exposure by eating aquatic species, and the contribution of PCBs in ingested water to the total dose of PCBs to birds is expected to be very low. Given that predictions in water are highly uncertain and PCBs have very low solubility, ingestion of waterborne PCBs was not considered.. This will be clarified in Section 4.3.1.7.
35	4.3.1.9	Results: The text states that the results of calculations using BAFs and regression models for invertebrates and plants were not tabulated, but were incorporated directly into the wildlife exposure model. For transparency, this particular part of the dose calculation shall be presented along with the corresponding soil/sediment exposure point concentration.	The BAFs and regression models are provided in Table 4-8; as described in the text, these are multiplied by the appropriate soil or sediment EPC to generate an estimate for plants and invertebrates. The resulting EPCs for invertebrates and plants will be added to Appendix C for transparency.
36	4.3.1.9	Results: Table 4-12 presents the final estimates of the daily ingestion rate of each $COPC_E$ for each receptor. We were not able to duplicate the values indicated for the raccoon. The report shall confirm/clarify the calculations. This may be related to uncertainty associated with the exposure areas assumed for the raccoon (i.e., see comment 9). 7/15/2012 Correction: i.e., see comment 67.	Please see response to comment 67, the table will be corrected to show exposure assumptions accurately, and this will correct the discrepancy. The table will be corrected.
37	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The linear regression models for each congener or homologue group from Elliott et al. (2001) were used to estimate egg concentrations for the blue heron, cormorant, and sandpiper. The regression equations are shown in Table 4-13. Levels of 2,3,7,8-TCDF were not linearly related for fish and egg concentrations ($p = 0.07$). The report shall discuss the uncertainty associated with the use of the Elliot, et al. (2001) model for this congener.	Additional information will be provided in Section 7.2.2.1 to describe uncertainty associated with the fish-to-egg model.
38	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The discussion on page 4-29 explains that for the fish-to-egg calculations, an individual sample of each medium was used to represent the CT and RM exposures. The sample selected was that with the $TEQ_{DF,B}$ concentration closest to the calculated CT or RM for the particular exposure unit. The report shall provide more discussion on why this calculation method was selected and the location, sample number, and congener and homologue concentrations of the individual samples selected for use. Additionally, this discussion states that it was considered overly conservative to use the CT and RM for each congener to estimate the concentrations of dioxins and furans in bird eggs. The report shall explain this statement.	Clarification will be provided in the discussion of model implementation in Section 4.3.2.1.2.

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39	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The results of the TEQ calculations using the regression models to estimate concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are shown in Table 4-15. For transparency, the report shall show the step-by-step calculation of the values in Table 4-15 for the combinations that follow. This would include presentation of the individual congener concentration EPCs (in food and sediment) as inputs to the calculation. a. Cormorant/TCFD/prey only/CT/TEF_{max}; b. Heron/PeCDD/prey + sediment/RM/TEFmin; c. Sandpiper/∑HxCDF/prey + sediment/CT /TEF_{min}.	An exhibit will be added detailing step-wise calculation of each of the requested combinations.
40	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): It appears that the TEF /TEQ values are missing for the heron and sandpiper (Table 4-15, background: prey + sediment). The report shall provide these values or explain why they were not presented.	Upstream data for shoreline sediments was inadvertently overlooked in calculation of background exposures of heron and sandpipers to dioxins and furans. The analysis will be revised and details added to Table 4-15. For estimation of egg PCBs, with consumption of prey and sediment, only the cormorant was evaluated because there is no background PCB data for shoreline sediment. The reviewer is referred to the last bullet in Section 4.3.2.3.
41	4.3.2.2.1	Overview of Literature Found (Estimating PCB Concentrations in Bird Eggs): The complete reference for Naito and Murata (2007) was not provided in the list of references. The report shall add this to the list of references. Additionally, the actual BMFs (biomagnification factors) in this paper were cited from other papers.	The appropriate citation will be added.
42	4.3.2.2.1	Overview of Literature Found (Estimating PCB Concentrations in Bird Eggs): The results of the TEQ calculations using the indicated BMFs (Table 4-16) to estimate PCB concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are shown in Table 4-17. For transparency, the report shall show the step-by-step calculation of the values in Table 4-17 for the combinations that follow. This would include presentation of the individual PCB congener concentration EPCs (in food and sediment) as inputs to the calculation. a. Cormorant/PCB l05/prey + sediment/CT; b. Cormorant/PCB126/background: prey + sediment/RM; c. Heron/PCB 077/background: prey/RM; d. Sandpiper/PCB 118/prey only/CT.	An exhibit will be added detailing step-wise calculation of each of the requested combinations and is compiled in the document following the figures.

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43	4.3.2.3	Egg Exposure Scenarios: Previous sections detail the approach for estimating egg TEQ_{DF} and TEQ_{P} concentrations using regression equations or BMFs applied to empirical fish tissue concentrations. This information is needed to evaluate potential risks to birds by comparing estimated TEQ concentrations in eggs to TRVs expressed as egg concentrations (wet weight). Exposure scenarios detailed here reflect an evaluation of egg concentrations resulting from combinations of prey (fish, crabs, or common rangia) and sediment. The report shall provide clarification regarding how egg tissue concentrations were estimated based on uptake from sediment, crabs, and common rangia. This is not clear.	Clarification was provided by reviewers during in a meeting with USEPA on July 18, 2012, as follows: There is uncertainty associated with using a model derived from data in which fish tissue is the independent variable in an application in which clam and crab tissue and sediment are combined to represent the independent variable. This uncertainty should be discussed in Section 7. Additional discussion will be added to Section 7 to address this uncertainty.
44	4.4.2	Derivation of Parameter Distributions: Table 4-19 displays the distribution characteristics for the various exposure parameters used in probabilistic risk analysis. The report shall discuss why any particular reference (e.g., DREBWQAT (1999) and Fernandes (2011)) was used here, and not in the initial dose calculations. Also, the report shall explain a triangular distribution.	The probabilistic risk assessment necessitates the use of not only a central tendency, which is consistent with the deterministic risk assessment references, but also a measure of variance and range, which are not contained in the deterministic risk assessment, hence the use of these additional references to provide these statistics. Clarification will be provided as requested.
45	5.3	Benthic Macro-invertebrate Communities: Notes f, h, and i are missing from Table 5-1. This table shall be revised to include these.	The table will be corrected. The same footnotes are missing from Table B-13, which is the duplicate benthic TRV table in Appendix B, and this table will be corrected as well.
46	5.3	Benthic Macro-invertebrate Communities: The marine chronic criterion for lead (Texas Surface Water Quality Standards (TSWQS), §307.6 (c)) of 5.3 ug/L shall be used for evaluating estimated pore water concentrations as this value is more conservative that the federal criterion. This is an ARAR (Applicable or Relevant and Appropriate Requirement).	The status of a benchmark as an ARAR is not a consideration in the selection of TRVs in a risk assessment. Moreover, the AWQC for lead was not needed for the risk assessment for benthic invertebrates, because values that could be used to evaluate risk to the benthic invertebrate community using the primary line of evidence, bulk sediment concentrations, were available. The surface water criterion for lead will be removed from Table 5-1 and from Table B-13.
47	5.3	Benthic Macro-invertebrate Communities: For the evaluation of reproductive risks for molluscs, the BERA used the paired NOAEC/ LOAEC (no-observed adverse effect concentration/lowest-observed adverse effect concentration) values of 2 and 10 ng TCDD/kg ww tissue, respectively, for delayed gonadogenesis in males (Wintermyer and Cooper (2007). An NOAEC of 2 ng TCDD/kg ww tissue is too high given that this concentration has been found to adversely affect early stages of oyster gametogenesis (Wintermyer and Cooper (2007) and veliger larval survival (Cooper and Wintermyer (2009). The report shall be revised to include the 2 ng TCDD/kg ww tissue concentration as the LOAEC, and a lower NOAEC shall be determined based on an appropriate literature value.	The TRV addressed by this comment is only appropriate for evaluation of risk to bivalve molluscs, as explained in Appendix B, because studies with other types of benthic macroinvertebrates have demonstrated that several macroinvertebrate taxa are not sensitive to 2,3,7,8-TCDD toxicity. The concentration of 2 ng TCDD/kg tissue will be considered the LOAEL in the revised BERA. There is no information to support identification of a corresponding NOAEL. However, we do not agree with Cooper and Wintermyer (2009), which cites Wintermyer and Cooper (2003) to support a conclusion that 2 ng/kg TCDD in eastern oysters (*Crassostrea virginica*) causes reduction in veliger larval survival. Wintermyer and Cooper (2003) placed their wild-caught test subjects in Newark Bay, in Arthur Kill of the Raritan Complex, and in a reference area (Sandy Hook), all in New Jersey. While Wintermyer and Cooper (2003) document the presence of TCDF and PCBs in adult oyster

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			tissue, they do not report on contamination of this area with metals, PAHs, and estrogenic compounds, all of which are common in urban estuaries and all of which could have affected the endpoints evaluated. Additional discussion of Wintermyer and Cooper (2003) will be provided in Appendix B and Section 5.3. The discussion of risk to bivalves in Section 6.2.3 will also be expanded to better explain uncertainties. Because the most technically robust evidence from this body of work is for a histological endpoint, and effects at the lowest concentrations were marginal in females and did not occur in males at the lowest dose, a NOAEC below 2 ng/kg tissue will not be proposed.
48	5.3	Benthic Macro-invertebrate Communities: Continuing with a discussion of the NOAEC/LOAEC values for molluscs, the referenced studies only dosed the molluscs with 2,3,7,8-TCDD, whereas the molluscs at the site are potentially exposed to all of the dioxin and furan congeners. Thus, site molluscs would have a greater exposure to total dioxins/furans overall. This compounds the uncertainty associated with the selected tissue residue endpoint for molluscs. The report shall evaluate/clarify this.	As described in Appendix B and in Attachment B2 to Appendix B of the RI/FS Work Plan, invertebrate cells do have aryl hydrocarbon receptor homologues, but these do not bind dioxin. Therefore, the toxicity of 2,3,7,8-TCDD is not necessarily an indication of toxicity of other 2,3,7,8-substituted dioxin and furan congeners as it is in vertebrates. Consequently, it is not appropriate to make any assumptions about compounding uncertainty. The draft BERA clearly states in Section 6.2.5 that exposure of bivalves to other dioxin and furan congeners cannot be interpreted due to a lack of toxicity information. Text will be added to Section 7, the uncertainty analysis, to highlight this data gap.
49	5.4	Fish: For nickel, the results of tests with marine fish were combined to determine a chronic TRV for nickel expressed as a concentration in water (3,600 ug/L; Table 5-2 and Table B-16). The marine chronic criterion for nickel (TSWQS, §307.6 (c)) of 13.1 ug/L shall be used. This is an ARAR.	The status of a benchmark as an ARAR is not a consideration in the selection of TRVs in a risk assessment. The nickel TRV for fish was well-considered and represents a range of marine fish species. The value presented in Tables 5-2 and B-11 is a conservative representation of no observed adverse effects concentrations for marine fish from the peer reviewed literature and USEPA's water quality criteria document for nickel. The derivation of the TRV is discussed in Section 3.9.1 of Appendix B. In light of the available information describing the actual toxicity of nickel to marine fish, it would be inappropriate to suggest that a value 200 times lower than the geometric mean of several NOAECs is a toxicity threshold. No change will be made. Tables 5-2 and B-11 will be corrected to show this value as a NOAEC.
50	5.4	Fish: The TRVs (NOAEL and LOAEL fish whole body concentrations) for Total PCBs are summarized in Tables 5-2 and B-11 and are discussed in Sections 2 .2 .1.1 and 2.2.1.2 of Appendix B. These TRVs are largely based on studies where fish were exposed to Aroclor 1254 and tissue was analyzed for Total PCBs. The report shall briefly discuss the uncertainty associated with the use of Aroclor toxicity data relative to the congener tissue data used for the BERA.	Additional discussion requested will be provided in Section 7 and in Appendix B.

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51	5.4	Fish: Regarding the TCDD TRV (from Steevens et. al. (2005)), our understanding is that the tissue residue TRV is based on concentrations in fish eggs and embryos rather than whole fish. The report shall clarify this. It appears that whole fish concentrations are used in the hazard quotient calculations (Section 6.3.4).	Clarification will be provided in the text of Section 5.4 and in Appendix B.
52	5.6	Birds and Mammals: The avian and mammalian TRVs for Total PCBs are summarized in Tables 5-3, 5-4, and B-11, and are discussed in Sections 2.2.3 and 2.2.4 of Appendix B. These TRVs are largely based on studies where birds or mammals were exposed to Aroclor 1254 in their diets. The report shall briefly discuss the uncertainty associated with the use of Aroclor 1254 (primarily) toxicity data relative to the total PCB (sum of Aroclors) tissue and sediment data used for the BERA.	Clarification will be provided in Section 7 and in the text of Appendix B.
53	5.6	Birds and Mammals: The report shall re-evaluate the calculated NOAEL and LOAEL values for the avian TRVs for barium. We were not able to duplicate the values indicated in Table 5-3 based on the text in Section 3.2.2 of Appendix B. The report shall also evaluate the indicated TRVs. Presumably this would be relevant for the SLERA for the area south of IH-10 because barium is not a $COPC_E$ for wildlife receptors for the area north of IH-10.	Inclusion of a discussion of a barium TRV for birds is a mistake, and related information will be removed from the report and Appendix B. If barium is a $COPC_E$ for birds in the south impoundment area, TRV calculations will be checked as requested.
54	6.2	Risks to Benthic Macro-invertebrate Communities: This discussion generally compares the various screening values with the bulk sediment or estimated pore water concentrations, indicates the number of exceedences, and plots the sample locations on a series of figures. This discussion shall be revised to indicate the concentrations (i.e., bulk sediment or estimated pore water) that exceeded the screening values.	The additional information requested will be provided in the maps cited in Section 6.2 which show results for those locations where concentrations exceed the TRV or screening value.
55	6.2.3	TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs: Potential risks associated with critical tissue residue in molluscs shall be reevaluated given the concerns regarding the selected tissue NOAEC/LOAEC values.	The discussion will be modified. Please see response to comment 47.
56	6.2.3	TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs: Absent confirmation sampling, it is unknown whether risks to molluscs in the vicinity of Transect 3 have been greatly reduced as a result of the TCRA. The report shall clarify this.	The reviewer is referred to the first sentence in the last paragraph of Section 6.2.3 which states: “It is not possible to evaluate post-TCRA risk to clams in the vicinity of Transect 3…” Clams were collected directly along the shoreline of the wastes from the northern impoundments, along Transect 3, as shown in Figure 4-1. This area is clearly within the TCRA footprint. The text will be modified to clarify the basis for the statement suggesting that the TCRA has affected exposure and risk to clams.
57	6.2.3/8.1	The conclusion that risks to bivalves are low in transects 3 and 5 based on the available data on clam tissue is not appropriate.	The document does not conclude that baseline (pre-TCRA) risks to molluscs from Transect 3 (adjacent to the impoundments) are “low.” The document acknowledges some reproductive risk to

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		If the assertion is that the TCRA has addressed the affected bivalves near the pits, monitoring post-TCRA will be necessary along with appropriate action levels in clam tissue.	*individual* bivalves in the area adjacent to the northern impoundments, stating in Section 6.2.3 that "individual clams from the area represented by Transect 3, assuming they are as sensitive as the oysters of Wintermyer and Cooper (2007), are at risk of reproductive impairment ". However, the assessment endpoint is "stable or increasing *populations* of bivalves." The possible effect on this assessment endpoint due to concentrations in bivalve tissue above the effects thresholds identified in samples from Transect 3 is unknown, because the spatial distribution of molluscs with those concentrations is unknown. However, data presented in the PSCR indicated that clam tissue concentrations of TCDD are somewhat correlated with sediment concentrations. Because very high concentrations of TCDD in sediments have limited distribution on the site, it is reasonable to conclude that effects on individuals are correspondingly limited, and that therefore, entire populations of bivalves in the site as a whole are not at risk. The text of Sections 6.2 and 8.1 will be revised to better convey the difference between risks to individuals and risks to the assessment endpoint. Concentrations of TCDD in three of five clam tissue samples from Transect 5 are below the lowest threshold of effects on molluscs. The effect indicated by the TRV is a histological abnormality, which is presumed to lead to some unspecified reproductive effect. The assessment endpoint addressed by clam tissue is "stable or increasing populations of bivalves." The conclusion that risks are low to bivalve populations because of slight exceedance of a histological effects threshold in less than half of the samples at Transect 5 is appropriate and will not be changed. Please note that the statements in the last paragraph of Section 6.2.3 do not assert that the TCRA has addressed risks to bivalves. It presents information that informs but does not attempt to resolve the post-TCRA risk condition.
58	6.2.5	Summary: Lines of Evidence for Benthic Macro-invertebrate Communities: The actual risk to populations of molluscs (based on tissue concentrations of dioxins/furans) is unknown. Additionally, consideration of potential risks to molluscs directly adjacent to the impoundment or elsewhere on the Site will be driven by the selected tissue NOAEC/LOAEC (see comments for Section 5.3). The report shall clarify this.	Please see response to comment 57.
59	6.3.1	Estimated Concentrations of Metals in Fish Diets Relative to TRVs: Hazard quotients for fish exposed to cadmium, copper, mercury, and zinc in foods and sediment are summarized in Table 6-3 and indicate that the LOAELs are not exceeded. These hazard quotients will be revisited based on the report revision in response to comment 8. 7/15/2012 Correction: i.e., see comment 65	Table 6-3 shows HQ calculated using NOAELs an LOAELs, all of which are below 1. The table and related conclusions will not be revised.
60	6.3.2	Estimated Concentrations in Surface Water Relative to TRVs: A hazard quotient of less than 0.1 was determined for fish exposed to nickel in surface water (Table 6-4). The hazard quotient will be above one using the chronic Texas criterion (see previous comment 39). The report shall be revised to include the chronic Texas criterion.	Please see response to comment 49.

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61	6.3.3	Total PCB Concentrations in Whole Fish Relative to the TRV for Fish: See previous comment 40 regarding the toxicity studies used to derive the fish whole body TRVs.	See response to comment 40.
62	6.3.5	Summary - Lines of Evidence for Fish: This discussion concludes that overall, risks to fish on the Site are negligible. This conclusion will be revisited based on the report revision in response to previous comments regarding the exposure concentrations (surface water), diet, and TRVs for fish.	Conclusions about risks to fish will not be revised on the basis of changes resulting from previous comments.
63	(7.)4.2.5	Datasets Used to Evaluate Exposure to Fish: The references for the killifish movement/home range were not provided in the reference section. The report shall provide the full references.	It appears that the commenter is referring to Section 4.2.5. The requested references will be added.
64	8.2	Characterization of Risks to Fish: The discussion summarizes that baseline risks to the assessment endpoints (stable or increasing populations of benthic omnivorous fish, benthic invertivorous fish, and benthic piscivorous fish on the Site) arc negligible. This conclusion will be revisited upon the report revision in response to previous comments regarding the exposure concentrations (surface water), diet, and TRVs for fish.	Please see the response to comment 62.
65	4.2.6 Corrected 7/15/2012	Results of Fish Exposure Assessment: The values in Table 4-6 shall be related with the exposure point concentrations in Appendix C, if applicable. If not applicable, the report shall explain how these weighted concentrations were derived and indicate where the data is summarized so this can be verified. Finally, the report shall clarify why is the total diet (last column in Table 4-6) simply the sum of each of the CT and RME values. Have the individual values for each food type already been modified by the proportion each food type represents in the diet?	Section 4.2.1 describes calculation of weighted fish diets. This section explains that the EPCs for each component of the fish diet, as expressed in Appendix C, are multiplied by the relative proportion of that item in the diet of the fish that is outlined in Table 4-2. The reviewer is referred to Equation 4-4, which provides the explanation of how the total diet is calculated; the values in Table 4-6 are summed to provide the total diet in the last column. This will be clarified by providing footnotes to Table 4-6 that describe this process.
66	8.6	Ecological Risk Assessment Conclusions: The overall risk assessment conclusions will be revisited after receipt of a revised BERA and accompanying responses to agency comments.	According to a discussion with USEPA and other reviewers on July 18, 2012, the draft final report will be submitted with a redline/strikeout of the text, to facilitate the USEPA's final review.
67	4.3 Corrected 7/15/2012	Exposure of Reptiles, Mammals, and Birds: Table 4-7 presents the exposure areas and assumptions for food/sediment/soil for various receptors. The exposure assumptions for the raccoon were a bit confusing. Presumably, concentrations in molluscs for the peninsula shoreline were used. It was not clear why this was not the case	This comment highlights two errors that will be corrected: In Table 4-7, the "Terrestrial Invertebrates" cell for Raccoon will be revised to state "BAFs from peninsula soils." The cell for "Benthic Invertebrates" will be revised to remove the reference to use of a BAF, because empirical tissue data were used in the exposure model for raccoon.

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		for small fish also since exposure point concentrations were presented for this subset in Appendix C. For terrestrial invertebrates and plants, it was unclear why concentrations were modeled from soil concentrations for soils north of IH-10 if soil ingestion was modeled for the entire peninsula. The report shall clarify/explain these issues.	In addition, empirical tissue data for clams and small fish were incorrectly used in the model from the entire aquatic area instead of restricting to the peninsula exposure area as correctly described by Table 4-7 and Figure 4-10. These calculations will be revised and dose and HQ tables will be updated as appropriate.
68	Table 4-5	Were there not 10 samples collected and analyzed from each FCA? The report shall clarify this.	Although 10 catfish fillet samples were collected from each fish collection area (FCA) for evaluation of human health risks, for ecological risk endpoints, a total of 10 whole catfish and 10 whole killifish were collected across the entire site. The sampling emphasized FCA2, which contains the area of the northern impoundments. This is the design described in the Sampling and Analysis Plan: Tissue Study (2010b). Clarification will be provided in Section 4.2.3.
69	Table 5-1	The report shall provide additional information supporting the assumption of dividing the LC_{50} by 10 results in a defensible estimation of the NOAEC. In this table, an uncertainty factor of 10 is applied to a LC_{50} resulting in a NOAEC and an EC_{50} yielding both a NOAEC and LOAEC. There is a disconnect in the logic in using this factor.	The reviewer is referred to Section 1.3 of Appendix B of the BERA, which describes the use of uncertainty factors. This information will be summarized in Section 7 of the BERA.
70	Table 5-1	This table has an incorrect reference for the TCDD value. The comment indicates that the range was derived from table B-5, but it should be Table B-4. The report shall be revised to correct this.	The correction will be made.
71	Table 5-2/ B-14	The TCDD value is described as a NOAEC; however, the source of this value indicates that it was the geometric mean of the NOER and LOER. The report shall either provide justification for the designation as a NOAEC or rename.	Clarification will be provided.
72	Table 5-2/ B-14/B-11	The source of the NOAEC and LOAEC for PCBs in fish is not clear. Although a summary of the studies used to derive these values is included in Section 2.2.1.1, 2.2.1.2, and Appendix B, it was not clear which of the studies were selected and which were not to calculate the NOAEC and LOAEC in this BERA. The report shall provide a table similar to B-4 for fish, and include only those studies used to calculate the TRVs.	A table will be prepared that shows the studies compiled to develop the PCB TRV for fish and clearly indicates which studies were selected for calculation of the NOAEL and LOAEL TRVs.
73	Figure 2-2	This figure combines the worker and trespasser receptor categories. Additional clarification/justification shall be provided for why these categories should be combined.	The CSM figures are for the site overall. Human receptors are not addressed by the BERA. Additional detail on the human health risk evaluation will be presented in the Baseline Human Health Risk Assessment.
Comments on Appendix E: Draft Screening-Level Ecological Risk Assessment, South Impoundment			
74	2.5	Assessment Endpoints: In Table E-3 (assessment endpoints), the assessment endpoint for mammals does not pair up with the selected receptor (pocket gopher) because it is an herbivorous mammal. The report shall include an omnivorous mammal (e.g., shrew, marsh rice rat, or armadillo) and revision of Table E-3.	The omnivorous mammal that will be added to the receptors evaluated by the ecological risk assessment for the south impoundment area is the opossum (*Didelphis virginiana*). The marsh rice rat is more appropriate for evaluation of aquatic exposures, and the shrew and armadillo would not be expected in habitats like that provided by the south impoundment area.

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75	3.2	Ecological Risk-Based Screening Methods: For the semi-volatile organic chemicals (SVOCs), footnotes shall be added to Table E-5 to indicate where the median value for the Site-specific background concentrations was used. Additionally the explanation for note c is unclear (also in Table E-6). The report shall clarify this.	The Site-specific background concentrations were inadvertently placed into a column suggesting that they are Texas median background concentrations. Also, the SVOC values should have been bold, indicating that they were used for screening in Table E-6, as stated in the existing footnotes. These errors will be corrected. Footnotes will be clarified.
Comments Dated February 7, 2013, on Draft-Final Baseline Ecological Risk Assessment Dated August 2012			
1	3.4.4	The full citation for the Shields 2012 reference (related to the brown pelican range) is not provided in the list of references. This citation shall be added.	The citation will be added to the reference list.
2	3.4.4	There is a typographic error in the last sentence of Section 3.4.4. The reference to Section 4.1.3.6 shall be revised to state Section 4.3.1.6.	The error will be corrected.
3	Table 4-8	The table was not revised to indicate that the raccoon's fish dose was modeled for the peninsula fish only as was stated in the response to comment number 67; it currently states "site wide". The table shall be revised to include this.	The table will be revised accordingly.
4	5.3, 6.2.3; Table B-4	Laboratory studies in Wintermyer and Cooper (2003) are relevant to these sections. In addition to the reproduction studies of the oysters transplanted to impacted field locations in New Jersey, Wintermyer and Cooper injected (laboratory) adult oysters with tritium-labeled TCDD, and these were strip spawned after 28 days of exposure. Eggs from each treatment group were fertilized with sperm from the corresponding treatment group. The nominal concentrations were 2.0 and 20 pg/g and the concentrations in tissue were reported as 0.966 and 27.7 pg/g TCDD. For both treatment groups, there was a reduction in the number of veliger larvae compared to controls. For the 2.0 pg/g treatment group, roughly half of the eggs were fertilized, and of those, there was 100% mortality within 48 hours. This lab study indicates a tissue LOAEC for impaired reproduction and reduced larval survival as low as 1 pg/g or 1 ng/kg. The BERA shall be revised to address this result. In an email from USEPA on April 19,2013: EPA agrees that the LOAEC value from Wintermeyer and Cooper (2003) included in the draft report is correct, and therefore retracts Comment #4 (Miller 2013, Pers. Comm.).	Although the original comment incorrectly interprets the dosing regime in the laboratory component of Wintermyer and Cooper (2003), the comment correctly states that the laboratory component of this study is relevant, and that the lower dose (2 ng/kg) in oyster tissue in the laboratory study resulted in reduced egg fertilization and reduced larval survival in oysters. The text of the BERA will be revised to address the potential for these effects in oysters with tissue concentrations of 2 ng/kg ww or greater. In addition, earlier text of responses require correction. For accuracy, the following parts of the response to earlier comment 47 (above) are retracted: Comment 47: However, we do not agree with Cooper and Wintermyer (2009), which cites Wintermyer and Cooper (2003) to support a conclusion that 2 ng/kg TCDD in eastern oysters (*Crassostrea virginica*) causes reduction in veliger larval survival … …Because the most technically robust evidence from this body of work is for a histological endpoint, and effects at the lowest concentrations were marginal in females and did not occur in males at the lowest dose... Similarly, the italic text in the following replaces the corresponding wording in the ninth sentence of the original response to earlier comment 57 (above): Comment 57: The effect indicated by the TRV *indicates* histological abnormalit*ies (Wintermyer and Cooper 2007), and reproductive effects including reduced fertilization success and reduced larval survival (Wintermyer and Cooper 2003).*
5	7.2.2.1	"PCBs" shall be removed from the title for this section because PCBs are not discussed there.	The section heading will be corrected.

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6	References	The link provided in the reference section for the U.S. Environmental Protection Agency paper on dioxin bioavailability (USEPA, 2010b) is incorrect and shall be revised.	The reference for USEPA (2010b) was inadvertently run in with USEPA (2012), making it appear that the link was incorrectly associated with the former reference. The two references will be separated with a line break, and a link will be will provided for the former reference.
7	References	The full citations for U.S. EPA (1986) and WHO (2001) in Table 5-1 were not carried forward to the reference list. These shall be added to the reference list.	These references will be added to the list.
8	Table B-12	This table lists the data used to develop fish tissue-based toxicity reference values (TRV). Suitable data references should be judged on criteria including sensitive life stage, chronic exposures, a protective endpoint (mortality is not very protective), species representative of the site receptors, and evaluating PCBs as a mixture (because that is what the site exposure would be). The following data sources are not appropriate for the reasons given, and shall not be included in the TRV derivation: a. Duke et al 1970 (only examined acute exposures); b. Lieb et al 1974 (used rainbow trout, a coldwater species not representative of Gulf of Mexico fish); c. Nestel and Budd 1975 (used rainbow trout, a coldwater species not representative of Gulf of Mexico fish); d. Mauck et al 1978 (used brook trout, a coldwater species not representative of Gulf of Mexico fish); e. Berlin et al 1981 (used lake trout, a coldwater species not representative of Gulf of Mexico fish); f. Mac and Seelye 1981 (used lake trout, a coldwater species not representative of Gulf of Mexico fish); and g. Powel et al 2003 (used chinook salmon, a coldwater species not representative of Gulf of Mexico fish). Instead, the following data sources shall be included in the TRV derivation: a. Orn et al 1998 (“The Impact on Reproduction of an Orally Administered Mixture of Selected PCBs in Zebrafish (Danio rerio)”); LOAEL 2.7 mg/kg. b. Westin et al 1983 (“Effects of Parental and Dietary PCBs on Survival, Growth, and Body Burdens of Larval Striped Bass”); NOAEL 3.1 mg/kg. In an email from USEPA on April 19,2013: Comment #8: EPA agrees that the NOAEL value from Westin et al. (1983) should be 4.4 mg/kg instead of 3.1 mg/kg, and therefore revises the comment to include the 4.4 mg/kg value in the TRV derivation. (Miller 2013, Pers.Comm.).	The studies of PCB toxicity to fish were selected based on criteria described in Section 1 of Appendix B. Moreover, the use of several salmonid species in calculation of the TRV is conservative because salmonids tend to be among the most sensitive fish taxa to many toxicants, including PCBs. Note that chinook salmon has both freshwater and marine life stages. Although we would not anticipate that PCB toxicity would be different in freshwater fish than in marine or estuarine fish, the changes will be made, as requested. However, there are important uncertainties associated with both studies that USEPA has decided to include. The effect on the final TRVs for total PCBs is to make them highly conservative. Orn et al. (1998) evaluated effects on individual organs in fish, requiring dissection and removal of the ovaries and liver. “Whole body” concentrations of total PCBs were measured after these organs were removed. Because PCBs concentrate in these organs, the authors acknowledge that the resulting “whole body” concentrations are biased low. Moreover, these authors used a selection of 20 PCB congeners, resulting in a mixture that is not commonly found in nature. The lack of representativeness of selected mixtures in PCB toxicity tests was a concern of USEPA in earlier comments 50 and 52 (above). Similarly, Westin et al. (1983) is also conservative. Westin et al. (1983) used just one treatment group, exposed the fish during a period of rapid growth, and found no effects. A concentration in fish associated with actual effects is not determined by this study, resulting in an unbounded NOAEL, a highly conservative representation of a TRV.
9	Appendix E	The table of contents shall be updated to reflect the additions of Sections 2.3.1.4 and 2.4.3.	The table of contents will be updated.

Note: Section, table, and figure numbers cited in comments dated March 15, 2012, and their respective responses, are those presented in the Draft Baseline Ecological Risk Assessment (BERA) dated March 15, 2012. Some of these numbers were subject to change when the Draft BERA was revised.

References:

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Integral, 2010a. Technical Memorandum on Bioaccumulation Modeling, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September 2010.

Integral, 2010b. Sampling and Analysis Plan: Tissue Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

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